PB84-126770

Assessment for Future Environmental Problems - Ocean Dumping

EG and G Environmental Consultants, Waltham, MA

Prepared for

Environmental Protection Agency, Washington, DC

Dec 83

U.S. Department of Commerce
National Technical Information Service
NTIS®

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.
EPA-600/6-84-001

2.

3. RECIPIENT'S ACCESSION NO.
PB84 126770

4. TITLE AND SUBTITLE
Assessment for Future Environmental Problems - Ocean Dumping

5. REPORT DATE
December 1983

6. PERFORMING ORGANIZATION CODE

7. AUTHOR(S)
Menzie, C.A., F. Babin, J. Cura, G. Mariani

8. PERFORMING ORGANIZATION REPORT NO.

9. PERFORMING ORGANIZATION NAME AND ADDRESS
EG&G Environmental Consultants
151 Bear Hill Road
Waltham, Massachusetts 02254

10. PROGRAM ELEMENT NO.

11. CONTRACT/GRANT NO.
68-02-3724

12. SPONSORING AGENCY NAME AND ADDRESS
Office of Exploratory Research
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC 20460

13. TYPE OF REPORT AND PERIOD COVERED
Final Report

14. SPONSORING AGENCY CODE
EPA/600/20

15. SUPPLEMENTARY NOTES

16. ABSTRACT

The objective of this report is to provide the U.S. Environmental Protection Agency's Office of Strategic Assessment and Special Studies with a technical basis for making decisions on research priorities and resource allocation as these relate to the question of ocean dumping. The program was organized into four tasks. First, historical trends in waste generation, disposal, and legislative and technological factors (as of 1982) were reviewed to indicate the likelihood that a particular waste type would be ocean dumped in the future. Second, the environmental implications of land-based alternatives were reviewed to provide background on the nature of risks associated with these alternatives. Third, the environmental implications of ocean disposal were reviewed for wastes and their constituents. Finally, based on the information generated in the first three tasks, recommendations are provided on future research needs and these are assigned either high, moderate, or low priority status.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Ocean Dumping Waste Management Research Planning Marine Pollution		

18. DISTRIBUTION STATEMENT
Released to Public

19. SECURITY CLASS *(This Report)*
Unclassified

20. SECURITY CLASS *(This page)*
Unclassified

21. NO. OF PAGES

22. PRICE

EPA Form 2220-1 (9-73)

PB84-125770

EPA-600/6-84-001
December 1983

ASSESSMENT FOR FUTURE ENVIRONMENTAL PROBLEMS - OCEAN DUMPING

by

C. A. Menzie
F. Babin
J. Cura
G. Mariani

EG&G Environmental Consultants
Waltham, Massachusetts 02254

Contract No. 68-02-3724

Project Officers

Stephen O. Wilson
Marvin Rogul
Office of Strategic Assessment and Special Studies
Office of Exploratory Research
Washington, DC 20460

OFFICE OF EXPLORATORY RESEARCH
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, DC 20460

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

NOTICE

This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

ACKNOWLEDGMENTS

This report was prepared by Dr. Charles A Menzie, Mr. Frederick Babin, Ms. Elaine Burke, Dr. Jerome Cura, Dr. Reginald Gillmor, Mr. Douglas Levin, Mr. Russell Wilder, and Dr. George Mariani of EG&G, Environmental Consultants. Sections relating to land-based disposal alternatives for sewage sludge were prepared, in part, by Drs. Warren Litsky and Chun-Kwun Wun of the University of Massachusetts at Amherst. Sections relating to land-based disposal alternatives for industrial wastes were prepared, in part, by Mr. William Potter of Princeton Environmental Engineering. Dr. Menzie coordinated the overall scientific effort and preparation of the report. Mr. Daniel Marini of EG&G provided administrative support. Dr. Steven Wilson of EPA served as Project Officer and guided the direction and focus of the program.

SUMMARY

The objective of this report is to provide the U.S. Environmental Protection Agency's Office of Strategic Assessment and Special Studies with a technical basis for making decisions on research priorities and resource allocation as these relate to the question of ocean dumping. The program was organized into four tasks. First, historical trends in waste generation, disposal, and legislative and technological factors (as of 1982) were reviewed to indicate the likelihood that a particular waste type would be ocean dumped in the future. Second, the environmental implications of land-based alternatives were reviewed to provide background on the nature of risks associated with these alternatives. Third, the environmental implications of ocean disposal were reviewed for wastes and their constituents. Finally, based on the information generated in the first three tasks, recommendations are provided on future research needs and these are assigned either high, moderate, or low priority status.

The following waste types were considered in this program: sewage sludge, dredged material, fly ash, flue-gas desulfurization (FGD) sludge, gypsum, acid iron waste, industrial sludges, and low-level radioactive waste (LLW). All of these waste types are expected to increase in volume in the future. With regard to the probability that these wastes will be ocean dumped in the future, the review of trends in disposal as well as legislative, economic and political factors suggests the following: sewage sludge (moderate-high probability), dredged material (high probability), fly ash (moderate probability), FGD sludge (moderate probability), gypsum (low-moderate probability), acid-iron waste (moderate-high probability), industrial sludges (low-moderate probability), LLW (moderate probability).

A review of land based alternatives indicated that no alternative is risk free, although some alternatives pose greater risks than others. The information presented underscores the need for a cross-media risk analysis approach to waste management. The following land-based waste disposal alternatives were examined: landfills and land burial (sewage sludge, dredged material, some industrial wastes, LLW); land spreading (sewage sludge, dredged material, some industrial wastes); incineration (sewage sludge, some industrial wastes); solidification (some industrial wastes); and deep well injection (some industrial wastes).

The fate and effects of ocean dumped wastes were reviewed in general and a more detailed review was provided on the fate and effects of specific classes of contaminants. Special attention was given to specific constituents because 1) major concerns regarding long-term effects of ocean dumping arise from questions related to the persistence and biological fate and effects of particular contaminants within the wastes; 2) many wastes have a number of contaminants in common (e.g., chlorinated hydrocarbons, heavy metals) and by focusing on these it is possible to formulate research recommendations that address one or more types of wastes; and 3) some wastes are highly variable and an evaluation of the effects of such wastes requires an understanding of the major contaminants of concern. The fates and effects of the following classes of waste constituents were reviewed: nutrients, metals, halogenated hydrocarbons, polynuclear aromatic hydrocarbons, and radionuclides.

Recommendations for future research are presented for twelve general categories of studies. Relative priorities (high, moderate, low) are assigned for each category of study for each waste type. For example, studies on uptake, storage, and depuration of metals are assigned a high priority status for industrial sludge but moderate priority status for other waste categories. The categories of studies assigned high priority status for two or more types of wastes include physical fate of particulate materials; uptake, storage, depuration of halogenated hydrocarbons; effects of sediment alterations on benthic larval settlement; site monitoring; inventory of waste constituents; and methodologies for cross-media risk assessment. Particular recommendations within these and other lower priority categories of studies are presented.

TABLE OF CONTENTS

Section Page

ACKNOWLEDGMENTS.. iii

SUMMARY.. iv

1 INTRODUCTION... 1-1

2 TREND ASSESSMENT....................................... 2-1

2.1 Quantities of Waste Generated........................ 2-1

2.1.1 Sewage Sludge...................................... 2-1
2.1.2 Dredged Material................................... 2-1
2.1.3 Coal Ash and Flue Gas Desulfurization Sludge from Coal Use by Electric Utilities.......... 2-3
2.1.4 Gypsum from the Basic Production in the Titanium Dioxide Industry.................... 2-9
2.1.5 Waste Acid Production in the Titanium Dioxide Industry............................. 2-11
2.1.6 Low-Level Radioactive Waste........................ 2-12
2.1.7 Seafood Processing Waste........................... 2-13

2.2 Current Distribution of Wastes Among Disposal Options.. 2-16

2.2.1 Sewage Sludge...................................... 2-16
2.2.2 Dredged Material................................... 2-16
2.2.3 Coal Ash from Coal Use by Electric Utilities. 2-16
2.2.4 Flue Gas Desulfurization Sludge.................... 2-17
2.2.5 Gypsum from the Basic Fertilizer Materials Industry.. 2-17
2.2.6 Waste Acid from the Titanium Dioxide Industry.. 2-17
2.2.7 Low-Level Radioactive Waste........................ 2-18
2.2.8 Seafood Processing Waste........................... 2-18
2.2.9 Industrial Hazardous Wastes........................ 2-18

2.3 Survey of U.S. EPA Regions........................... 2-19

2.3.1 Sewage Sludge...................................... 2-19
2.3.2 Dredged Material................................... 2-21
2.3.3 Fly Ash.. 2-21
2.3.4 RCRA-Driven Chemical Waste and Pretreatment Requirements....................................... 2-21

Section

Page

2.3.5 Gypsum.. 2-21
2.3.6 Low-Level Radioactive Waste.................... 2-21
2.3.7 Ocean Incineration.............................. 2-22
2.3.8 Construction/Demolition Debris.................. 2-22
2.3.9 Seafood Processing Waste........................ 2-22
2.3.10 Waste Liquor................................... 2-22
2.3.11 Secondary Sludge from Pulp and Paper........... 2-22
2.3.12 Phosphate Mining............................... 2-22
2.3.13 Drilling Muds.................................. 2-23
2.3.14 Deep Ocean Mining.............................. 2-23
2.3.15 Tires.. 2-23
2.3.16 Vessels.. 2-23

2.4 Survey of Municipalities............................ 2-23
2.5 Survey of Industries 2-24

2.5.1 Pulp and Paper.................................. 2-24
2.5.2 Chemical.. 2-25
2.5.3 Pharmaceutical.................................. 2-25

2.6 Regulatory Factors.................................. 2-25

2.6.1 The Marine Protection, Research, and Sanctuaries Act.............................. 2-25
2.6.2 The Federal Water Pollution Control Act and National Pollutant Discharge Elimination System.. 2-26
2.6.3 Pretreatment.................................... 2-26
2.6.4 Waivers from Secondary Treatment................ 2-27
2.6.5 Ocean Discharge Criteria........................ 2-27
2.6.6 Resource Conservation and Recovery Act.......... 2-27
2.6.7 Safe Drinking Water Act......................... 2-28
2.6.8 Low-Level Radioactive Waste Policy Act.......... 2-28
2.6.9 State and Local Regulatory Factors.............. 2-28

2.7 Costs of Alternative Waste Disposal Methods......... 2-29

2.7.1 Sewage Sludge................................... 2-29
2.7.2 Dredged Material................................ 2-29
2.7.3 Coal Ash and Flue Gas Desulfurization Sludge.   2-31
2.7.4 Waste Gypsum.................................... 2-31
2.7.5 Waste Acids..................................... 2-32
2.7.6 Low-Level Radioactive Waste..................... 2-32
2.7.7 Seafood Processing Wastes....................... 2-33
2.7.8 Industrial Hazardous Wastes..................... 2-33

3 ENVIRONMENTAL AND PUBLIC HEALTH IMPLICATIONS OF LAND-BASED DISPOSAL ALTERNATIVES............................ 3-1

3.1 Landfills and Land Burial........................... 3-1

Section Page

3.1.1 Sanitary Landfills.......................... 3-1
3.1.2 Hazardous Waste Landfills.................... 3-4
3.1.3 Shallow Land Burial and Low-Level Radioactive Waste.................................... 3-5

3.2 Land Spreading..................................... 3-9

3.2.1 Land Spreading of Sewage Sludge.............. 3-9
3.2.2 Land Spreading of Industrial Wastes.......... 3-18

3.3 Incineration....................................... 3-18

3.3.1 Incineration of Sewage Sludge................ 3-18
3.3.2 Incineration of Hazardous Organic Wastes..... 3-19

3.4 Solidification..................................... 3-22
3.5 Deep-Well Injection................................ 3-22

4 ENVIRONMENTAL AND PUBLIC HEALTH IMPLICATIONS OF OCEAN DUMPING... 4-1

4.1 Dispersion Processes for Dumping Methods........... 4-1

4.1.1 Liquids from Ships........................... 4-2
4.1.2 Sludges, Dredge Materials, and Precipitates.. 4-5
4.1.3 Slow Bottom Releases......................... 4-9

4.2 General Effects of Ocean Dumping................... 4-10
4.3 Fate and Effects of Waste Constituents............. 4-12

4.3.1 Nutrient/Organic Enrichment.................. 4-12
4.3.2 Metals....................................... 4-16
4.3.3 Polycyclic Aromatic Hydrocarbons............. 4-20
4.3.4 Halogenated Hydrocarbons..................... 4-22
4.3.5 Low-level Radioactive Waste.................. 4-25

5 RECOMMENDATIONS FOR FUTURE RESEARCH.................... 5-1

5.1 Physical Fate of Dissolved Constituents............ 5-1
5.2 Physical Fate of Particulate Materials............. 5-4
5.3 Effects of Nutrient Enrichment..................... 5-5
5.4 Uptake, Storage, and Depuration of Metals.......... 5-6
5.5 Uptake, Storage, and Depuration of PAHs............ 5-7
5.6 Uptake, Storage, and Depuration of Halogenated Hydrocarbons....................................... 5-8
5.7 Uptake, Storage, and Transport of Radionuclides.... 5-9
5.8 Effects of Sediment Alterations of Benthic Larval Settlement................................. 5-10
5.9 Toxicity Studies................................... 5-10

Section Page

5.10 Site Monitoring.................................... 5-11
5.11 Inventory of Waste Constituents..................... 5-11
5.12 Cross-Media Risk Assessment......................... 5-11

6 REFERENCES... 6-1

Appendix

A NATIONAL MARINE POLLUTION CONCERNS REGARDING MARINE WASTE DISPOSAL FROM THE NATIONAL MARINE POLLUTION PROGRAM PLAN: FEDERAL PLAN FOR OCEAN POLLUTION RESEARCH, DEVELOPMENT, AND MONITORING FISCAL YEARS 1981-1985.................... A-1

LIST OF ILLUSTRATIONS

Figure		Page
2-1	Volume of sewage sludge produced in the U.S., with projections.	2-2
2-2	Volume of dredged material produced in the U.S. 1974-1981. .	2-4
2-3	Quantity of coal ash produced in the United States by electric utilities 1975-1981	2-6
2-4	Quantity of flue gas desulfurization sludge produced by electric utilities in the United States 1975-1980.	2-7
2-5	Quantity of gypsum produced as a by-product in the basic fertilizer industry 1976-1981 with projections	2-10
2-6	Volume of low-level radioactive waste produced and disposed of at commercial facilities in the United States 1962-1979 with projections	2-14
2-7	Seafood waste quantities	2-15
2-8	Map of U.S. EPA standard federal regions and regional offices.	2-19
4-1	Lateral diffusivity K_a as a function of horizontal scale for a variety of areas and experimental conditions	4-3
4-2	Bathymetric change at the dredge spoil dump site	4-7
4-3	A schematic illustration of the physical fate of solids in wastes disposed of at sea.	4-8
4-4	Relative contribution of various sources of metals to the New York Bight	4-17
4-5	A summary of the major steps in metabolism of DDT.	4-26
4-6	Possible routes of radionuclide transport in the marine environment.	4-29

LIST OF TABLES

Table		Page
1-1	Materials that require special care or permits for ocean dumping.	1-3
1-2	Materials generally prohibited from ocean dumping. . .	1-3
2-1	Volume of dredged material produced in the U.S. . . .	2-3
2-2	Coal use in the United States by electric utilities 1975-1981.	2-5
2-3	Projections of ash and FGD sludge production for electric utilities burning coal.	2-9
2-4	Basic fertilizer materials industry production capacity phosphate-based materials.	2-11
2-5	Waste acid production in the titanium dioxide industry 1972-1974.	2-12
2-6	Sewage sludge disposal methods and relative use. . . .	2-16
2-7	Coal ash disposal options and their relative use . . .	2-17
2-8	Results of telephone survey of U.S. EPA regional ocean dumping coordinators.	2-20
2-9	Cost and benefit items for consideration	2-30
2-10	Cost estimates for disposal of sewage sludge by various options	2-31
2-11	Cost estimates for disposal of FGD sludge by various options	2-31
2-12	Cost estimates for disposal of industrial hazardous wastes by various options	2-33
3-1	Concentration range of pollutants in leachate from typical domestic refuse	3-3
3-2	Weighted comparative analysis for alternatives	3-8

Table Page

3-3 Major organisms of health concern that may be present in sewage from U.S. communities. 3-12

3-4 Parasites found in sludge samples from 27 municipal plants in southern United States 3-14

3-5 Reported pathogen survival times 3-15

3-6 Particulate emissions from the furnace of a modern waterwall incinerator 3-20

3-7 Properties of particulates leaving furnaces. 3-20

3-8 Typical wastewater analyses. 3-21

4-1 General effects of ocean dumping various kinds of wastes. 4-11

4-2 Characteristics and constituents of selected classes of wastes which may be ocean dumped. 4-13

4-3 Major radionuclides in low level radioactive waste . . . 4-27

4-4 Total amounts of LLW dumped at the northeast Atlantic site and corresponding IAEA release rate limits. 4-31

5-1 Categories of studies and their relative priorities with regard to ocean dumping of various waste types. . . 5-2

1. INTRODUCTION

The objective of this report is to provide the U.S. Environmental Protection Agency's (EPA's) Office of Strategic Assessment and Special Studies (OSASS) with a technical basis for decisions on research priorities and resource allocation as these relate to the question of ocean dumping. The need for such information is clear. A number of factors (e.g., legislative, sociological, and technological) will have to be taken into account when considering various alternatives for waste disposal. For example, under the Marine Protection, Research, and Sanctuaries Act, ocean disposal of various wastes is being phased out. However, at the time of this writing (1982) the Act is up for reauthorization and ocean disposal is the subject of extensive congressional hearings. Other regulations will continue to result in changes in composition and amounts of various waste types. As an example, implementation of the Clean Water Act resulted in construction of numerous secondary sewage treatment plants and has contributed to an increase in sewage sludge volume; population increases will further contribute to an increase in sludge. In its report to Congress and the President, the National Advisory Committee on Oceans and Atmospheres calls the legislated ban on ocean dumping of sludge "unreasonable" and suggests that some of the earlier concerns about ocean waste disposal were "overstated."

Implementation of the Resource Conservation and Recovery Act has focused and will continue to focus attention on various wastes and alternatives for their disposal; for some, ocean dumping may be a better alternative than land-based disposal options. As increased emphasis is given to the use of coal over oil for power generation, there will be an increase in coal ash and slag; the U.S. may wish to consider ocean dumping of this waste as is presently being done by the United Kingdom. For certain wastes there has been increased international (through the Intergovernmental Maritime Consultative Organization [IMCO]) and national interest in incineration at sea.

The U.S. discontinued ocean dumping of low level radioactive waste (LLW) in 1970. However, other countries (e.g., England, Belgium, France, Switzerland, and the Netherlands) have continued to dispose of LLW in the northeast Atlantic under the review of the Organisation for Economic Cooperative Development (OECD) Nuclear Energy Agency (NEA); Japan, which is also a member of NEA, is considering ocean disposal of LLW in the Pacific. Although the London Convention (and U.S. regulations) prohibits ocean disposal of "high level radioactive waste," the convention (and U.S. regulations) also calls for the issuance of permits for the disposal of radioactive matter other than "high level." In issuing such permits, the contracting parties should take full account of the recommendations of the competent international body in this field, at present the International Atomic Energy Agency. There is still interest in the United States for future ocean disposal of LLW.

Among the various factors related to future consideration of ocean dumping as a waste disposal alternative is the assimilative capacity of the ocean for pollutants. The philosophy is that the oceans can accept or assimilate certain pollutants at some rate of disposal in particular areas. Determination of these levels through technical means (e.g., studies of fates and effects combined with appropriate modeling techniques) then provides a basis for permitting or restricting discharges. Preliminary efforts to assess the assimilative capacity of the ocean were made at the Crystal Mountain Workshop held by NOAA in December 1979. A panel on sources attempted to identify anticipated amounts and types of industrial, agricultural, and domestic wastes for which disposal in the ocean might be considered. Six other panels studied the assimilative capacities of water bodies. Two of those considered problems related to estuaries and the coastal and open ocean. The four remaining panels studied the site specific problems of Dumpsite 106, Puget Sound, the New York Bight, and the Southern California Bight. While the workshop indicated that evaluating the assimilative capacity of the ocean was useful in determining limits on discharges, the inadequacies of present-day models to predict impacts to ecosystems, with respect to long-term, low-level effects, was well recognized. The long-term chronic effect of marine pollution also was identified as a primary concern in the Global 2000 Report to the U.S. President prepared by the Council on Environmental Quality.

An initial task in our program was to determine classes or types of wastes which might be considered for future ocean dumping. Several criteria were used to select the wastes considered in this study. These criteria were:

1. The study included wastes permitted for ocean dumping by the London Convention (Table 1-1) but not those prohibited (Table 1-2). Therefore, wastes such as organochlorines, which are prohibited from ocean dumping but which may be incinerated at sea, are not discussed. The incineration at sea alternative, which would be used for wastes generally prohibited from ocean dumping, is not discussed within this report.

2. The study considered classes of wastes that, if disposed of at sea, would be typically "ocean dumped" (e.g., sewage sludge) as opposed to "discharged" (e.g., effluent from coastal treatment plants). The former classes of wastes and disposal methods are presently regulated under the Ocean Dumping Permit Program, the latter under the NPDES Permit Program.

3. The study included nonhazardous wastes that, because of their high volume and characteristics, may pose problems for land disposal (e.g., sewage sludge and fly ash); the study did not consider low volume nonhazardous wastes.

4. The study included wastes defined as hazardous under EPA's hazardous waste criteria but which are not excluded from ocean dumping by the London Convention (Table 1-2).

These criteria led to the selection of three broad categories of wastes consisting of a number of different waste types. These include:

A. High Volume Nonhazardous Wastes

- sewage sludge
- dredged material
- fly ash

TABLE 1-1. MATERIALS THAT REQUIRE SPECIAL CARE OR PERMITS FOR OCEAN DUMPING. (ANNEX II OF LONDON CONVENTION)

---

1. Wastes containing significant amounts of the matters listed below:

   arsenic, lead, copper, zinc } and their compounds
   organosilicon compounds
   cyanides
   fluorides
   pesticides and their by-products not covered in Annex I (see Table 112).

2. In the issue of permits for the dumping of large quantities of acids and alkalies, consideration shall be given to the possible presence in such wastes of the substances listed in Number 1 and to the following additional substances:

   beryllium, chromium, nickel, vanadium } and their compounds

3. Containers, scrap metal, and other bulky wastes liable to sink to the sea bottom which may present a serious obstacle to fishing or navigation.

4. Radioactive wastes or other radioactive matter not included in Annex I. In the issue of permits for the dumping of this matter, the recommendations of the competent agencies in this field should be considered.

5. Dredge spoils.

6. Substances that, though of a non-toxic nature, may become harmful due to the quantities in which they are dumped, or that are liable to seriously reduce amenities.

7. Titanium dioxide (specified by Council of European Communities).

TABLE 1-2. MATERIALS GENERALLY PROHIBITED FROM OCEAN DUMPING.[a,b] (ANNEX I OF LONDON CONVENTION)

1. Organohalogen compounds and compounds that may form such substances in the marine environment, excluding those that are non-toxic, or that are rapidly converted in the sea into substances that are biologically harmless.
2. Organosilicon compounds and compounds that may form such substances in the marine environment, excluding those that are non-toxic, or that are rapidly converted in the sea into substances that are biologically harmless.
3. Substances that are likely to be carcinogenic under conditions of disposal.
4. Mercury and mercury compounds.
5. Cadmium and cadmium compounds.
6. Persistent plastics and other persistent synthetic materials that may float or may remain in suspension in the sea so as to interfere materially with fishing, navigation, or other legitimate uses of the sea.
7. Crude oil, fuel oil, heavy diesel oil, lubricating oils, hydraulic fluids, and any mixtures containing any of these, taken on board for the purpose of dumping.
8. High-level radioactive wastes or other high-level radioactive matter, defined on public health, biological, or other grounds, by the competent international body in this field, at present the International Atomic Energy Agency, as unsuitable for dumping at sea.
9. Materials in whatever form (e.g., solids, liquids, semiliquids, gases, or in a living state) produced for biological and chemical warfare.

---

a. This table does not apply to substances that are rapidly rendered harmless by physical, chemical, or biological processes in the sea, provided they do not:

   (i) make edible marine organisms unpalatable, or
   (ii) endanger human health or that of domestic animals.

b. This table does not apply to wastes or other materials (e.g., sewage sludges and dredged spoils) containing the matters referred to in Numbers 1-7 above as trace contaminants.

- flue gas desulfurization (FGD) sludge
- gypsum waste
- seafood wastes

B. Low-Level Radioactive Wastes (LLW)

C. EPA Hazardous Wastes Not Excluded by the London Convention

- These wastes include those identified by the EPA as hazardous under the Resource Conservation and Recovery Act. Although there may be a number of waste types within this category, the only one for which we have a reasonable data base for the purposes of this assessment is waste acid.

The Program was organized into four tasks. First, historical trends in waste generation, distribution among alternative disposal methods, and disposal costs were reviewed, and legislative, technological, and economic factors that may influence these were examined. This task served to indicate the likelihood that a particular waste type would be ocean dumped in the future. Second, the environmental implications of land-based alternatives were reviewed to provide background on the nature of risks associated with these alternatives. These risks would have to be considered during formal cross-media risk analyses. Third, the environmental implications of ocean disposal were reviewed for the selected wastes and their constituents. This review aided in identifying technical areas where there was a lack of information, research, or analytical techniques for evaluating environmental impacts associated with ocean disposal. The final task was to provide recommendations for future research.

2. TREND ASSESSMENT

2.1 QUANTITIES OF WASTE GENERATED

2.1.1 Sewage Sludge

Quantities of sewage sludge produced in the U.S. have been increasing steadily as the nation's population has grown, as the proportion of the population served by municipal wastewater treatment plants has grown, and as more efficient treatment methods were developed and applied at individual treatment plants. Figure 2-1 presents estimates of past and possible future quantities of sludge generated in the U.S. beginning in 1972 as compiled by several investigators (NRC, 1977; Booz, Allen, and Hamilton, 1982; NACOA, 1981; Litsky [citing others], 1982). For the most part, these estimates have been reasonably consistent, although the Booz, Allen, and Hamilton (BAH) estimates for 1980 and projected 2000 exceed the other estimates by a factor of two. The projections compiled by NRC show an increase of 100% (3.44 to 6.88 million metric tons dry weight) between 1972 and 1980. By 1990, both NRC and NACOA anticipate that approximately 10 million metric tons of sewage sludge solids will be produced, with this quantity rising to 12.2 million metric tons in the year 2000. The BAH projection for the year 2000 is 18.9 million metric tons.

As pointed out in the NRC report, there are significant regional differences in quantity projections, with the Atlantic and southern Pacific coastal areas showing estimated 200% and 90% increases, respectively. These two regions are projected to generate more than 40% of the U.S. total sludge in 1990.

To gain additional insight into future sludge quantities, a telephone survey was conducted of selected municipalities on the east and west coasts. In general, sludge volumes are expected to increase due primarily to expanded service and the introduction of improved solids removal technology. The anticipated increases for individual systems ranged from 0% (South Essex Sewage District, Massachusetts) to 60% (Nassau County, New York).

2.1.2 Dredged Material

Dredged material results from the dredging of harbors, rivers, and channels to maintain or improve navigability. Particular dredging projects are generally defined as either maintenance or improvement projects, undertaken by the U.S. Army Corps of Engineers (COE) or by entities other than the COE. Maintenance projects involve the periodic redredging of navigation channels to authorized depths. These projects result from the deposition of sediments in navigable areas and are influenced by oceanographic and meteorological conditions in particular geographic areas. Improvement projects involve the deepening and/or

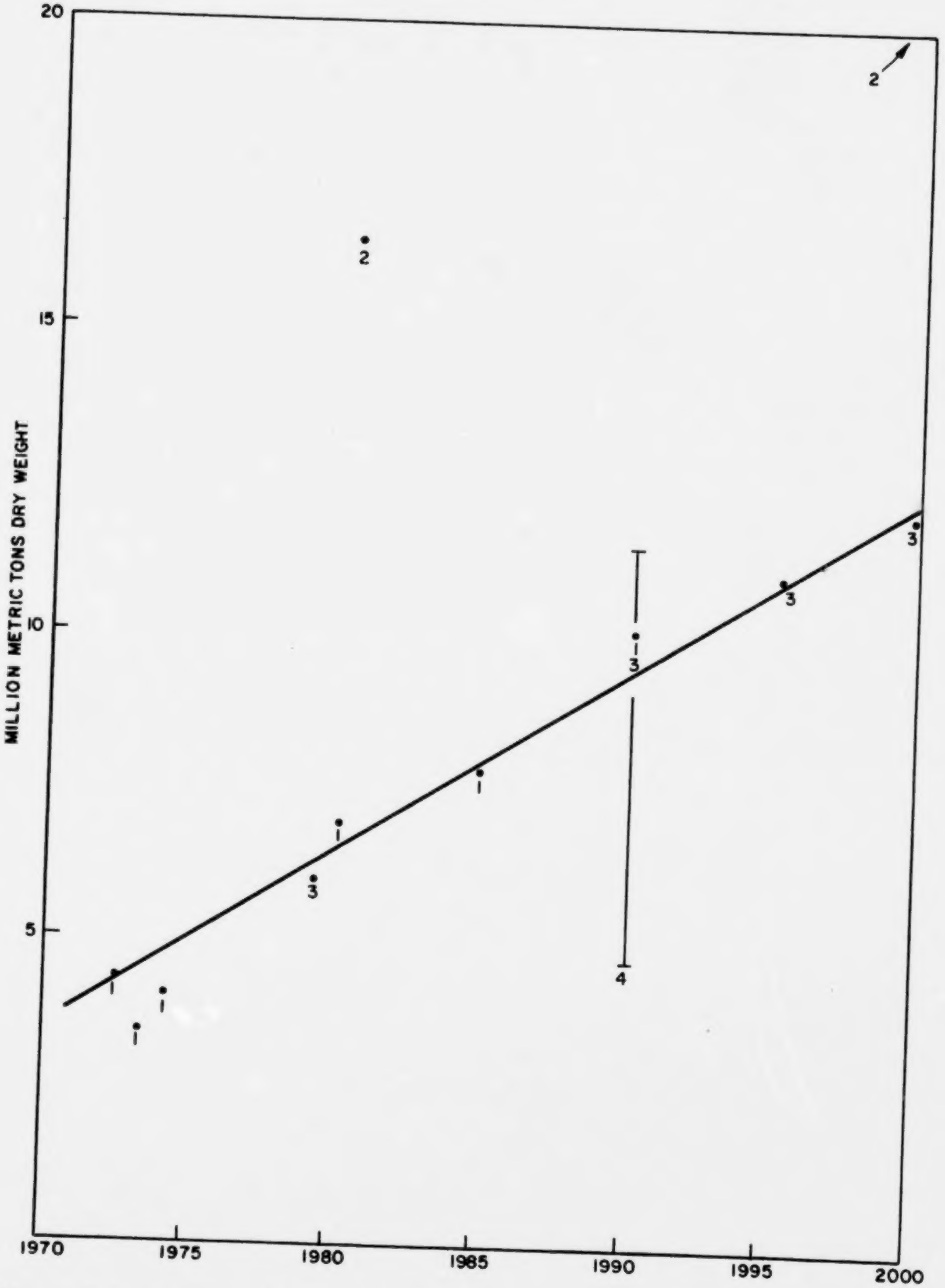


Figure 2-1. Volume of sewage sludge produced in the U.S., with projections. Sources include 1) NRC (1977); 2) BAH (1982); 3) NACOA (1981); 4) Litsky (1982).

widening of channels, anchorages, or turning basins conducted in existing or new areas.

Between 1974 and 1981, a total of 2493 million cubic yards of material were dredged in the U.S. for an annual average of 311.6 million cubic yards. The range in annual quantities was 386 to 272 million cubic yards in 1974 and 1979, respectively. Table 2-1 and Figure 2-2 summarize dredging activity in the U.S. over this 8-year period. Projecting future quantities of dredged materials is difficult because of the dependency of dredging projects on federally authorized funding programs and of dredging needs on weather conditions. It is probably safe to assume, however, that dredging activity will continue much as it has in the past. Increasing attention to deepwater port development may lead to large volumes of materials in need of disposal.

TABLE 2-1. VOLUME OF DREDGED MATERIAL PRODUCED IN THE U.S.

1974-1981 (10^6 yd^3)

1974	386
1975	332
1976	301
1977	298
1978	280
1979	272
1980	289
1981	335

2.1.3 Coal Ash and Flue Gas Desulfurization Sludge from Coal Use by Electric Utilities

According to the U.S. Department of Energy (DOE, 1981), electric utilities in the U.S. produced 2286 billion kilowatt hours (kwh) of electric energy in 1980. This figure represents a 1.7% increase over the 1979 figure of 2247 billion kwh, and a 19% increase over the 1975 level of 1918 kwh. The use of coal in electricity generation has also increased steadily over this period, in both absolute (approximately a 40% increase between 1975 and 1980) and relative (14% increase between 1975 and 1980) terms. In 1975, coal was utilized as the basic fuel in producing 44.5% of the total electricity output of the U.S. During 1980, coal accounted for nearly 51% of the basic fuel used in electricity generation.

Coal use on the part of electric utilities is expected to continue to increase in the future. The U.S. Congress Office of Technology Assessment (OTA, 1979) has reviewed a number of alternative projections and considers 775 million tons and 1410 million tons to be maximum utility coal consumption estimates for 1985 and 2000, respectively. The same report also recognizes that continued slow growth in electricity demand is likely, resulting in more conservative estimates of coal use of 725 and 1275 million tons in 1985 and 2000, respectively. This growth will come about as existing plants which burn oil and/or gas are converted to coal combustion systems, and as new coal burning capacity is installed.

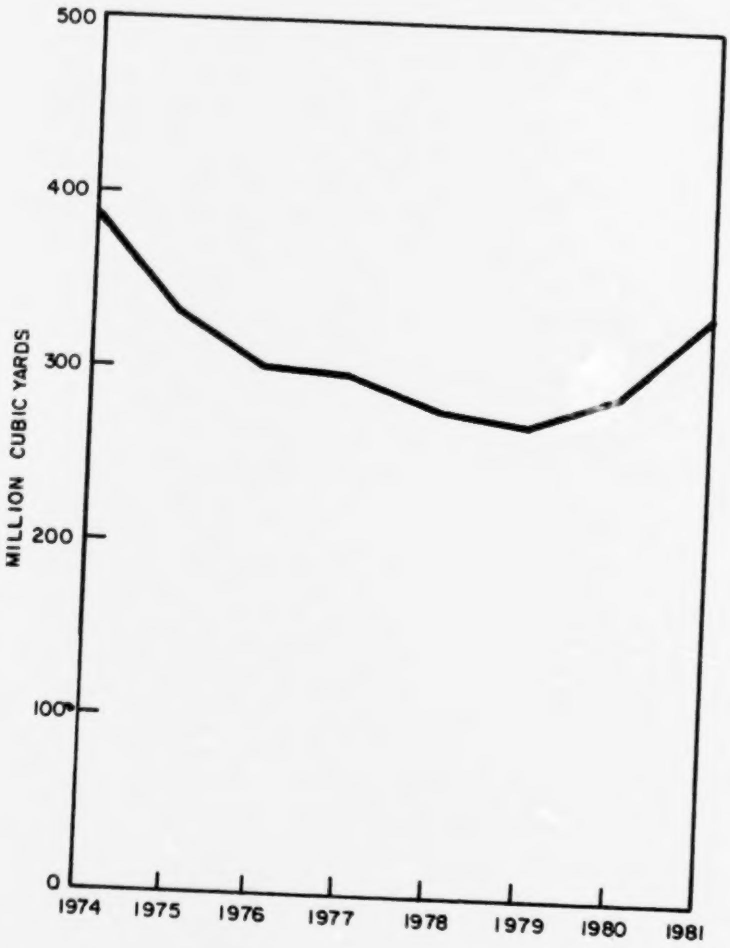


Figure 2-2. Volume of dredged material produced in the U.S. 1974-1981.

Coal combustion produces two major wastes of concern due to their composition and volume - ash and flue gas desulfurization (FGD) sludge. In 1975, nearly 70% of the coal consumed in the U.S. was burned to produce electricity (OTA, 1979). This represents a significant change from 1950 when industrial and residential/commercial use of coal accounted for approximately 80% of coal consumption. Since the mid-1950s, the consumption of coal in the U.S. has been increasing steadily while the electric utility share of this consumption has grown from 55% to 80%. Clearly, electric utilities represent the key sector when considering future disposal options for the by-products ash and FGD sludge.

Figures 2-3 and 2-4 depict waste generation characteristics for the 1975-1981 period by electric utilities in the U.S. The quantities presented are those associated with "coastal" and "non-coastal" states as well as for the U.S., because when considering the potential of ocean disposal of coal combustion waste products, generators located in states in coastal regions will probably be more likely to utilize this option. The ash and FGD sludge estimates are dry weight figures, and were derived by considering the quantity of coal consumed, the ash and sulfur content of the coal, and typical ash and sulfur recovery rates of existing pollution control technology. The FGD sludge estimates also incorporate data describing the capacity of FGD systems in place in each year (DOE, 1978, 1979, 1980) and include a range in estimates based on the type of technology employed (limestone or double alkali systems). As reported by the U.S. Nuclear Regulatory Commission (USNRC, 1977), the limestone and double alkali techniques bracket the range of FGD sludge generation rates for available air emissions sulfur recovery systems.

An examination of Table 2-2 shows that electric utility coal use has been increasing as discussed above. As a result, quantities of ash generated (Figure 2-3) have also been rising steadily from nearly 50 million tons in 1975 to approximately 68 million tons in 1981. The increase has been most pronounced in coastal states, with ash quantities increasing from 26 million tons in 1979 to an estimated 31 million tons in 1981. Because ash recovery systems are typically capable of capturing over 99% of the ash generated in the form of fly ash, bottom ash, or slag (USNRC, 1977), there is little potential for ash quantities to increase further as a rsult of improved ash recovery technology. Rather, future ash generation will change as the use of coal by electric utilities increases, and as the characteristics of coal change.

TABLE 2-2. COAL USE IN THE UNITED STATES BY ELECTRIC UTILITIES 1975-1981. (1000 TONS)

	1975	1976	1977	1978	1979	1980	1981
Coastal states	209,979	213,780	233,751	218,028	206,192	235,542	265,156
Non-coastal states	222,453	242,098	258,189	299,688	350,353	358,456	347,767
Total U.S.	432,432	455,878	491,940	517,716	556,545	593,998	612,923

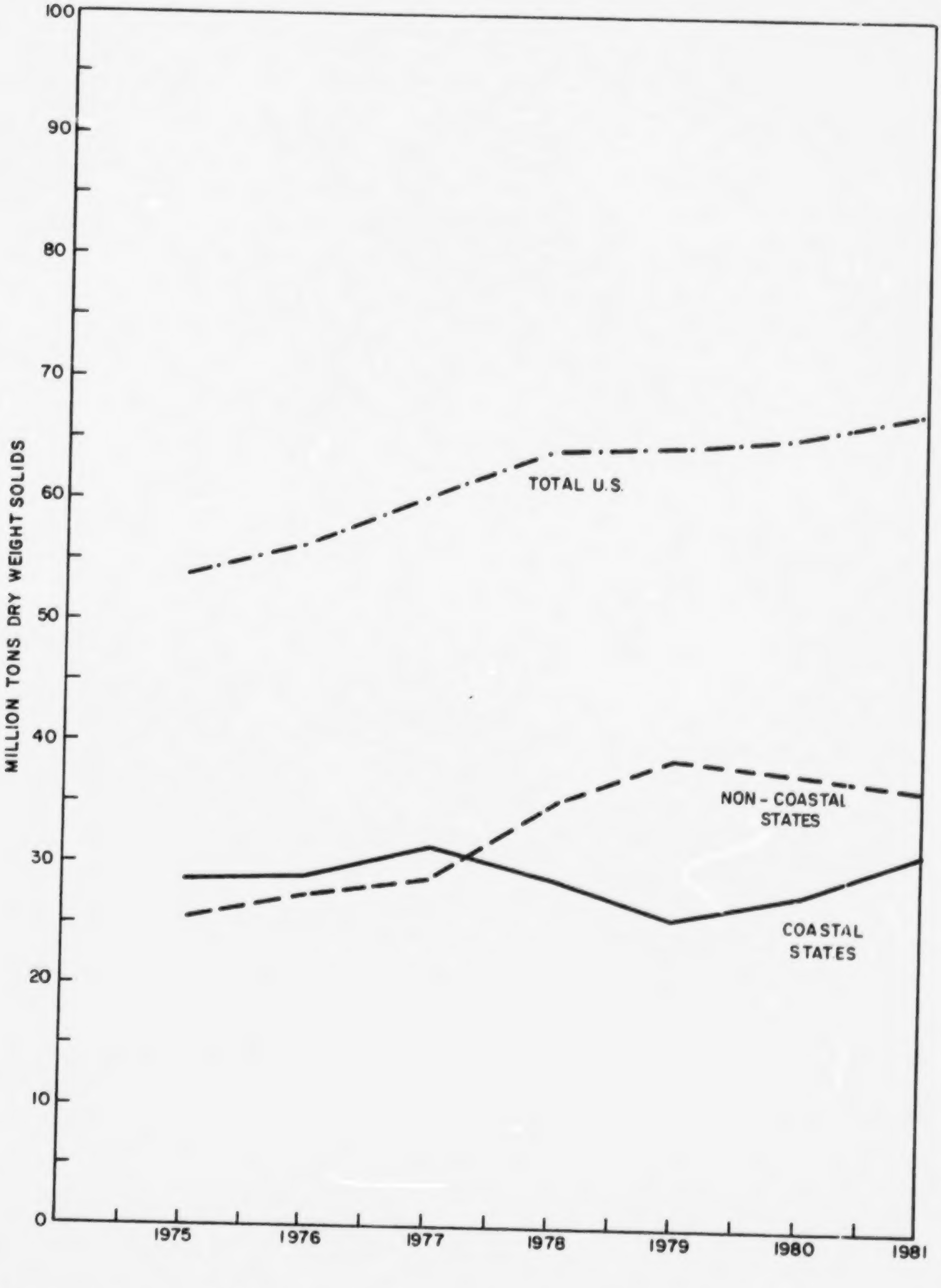


Figure 2-3. Quantity of coal ash produced in the United States by electric utilities 1975-1981.

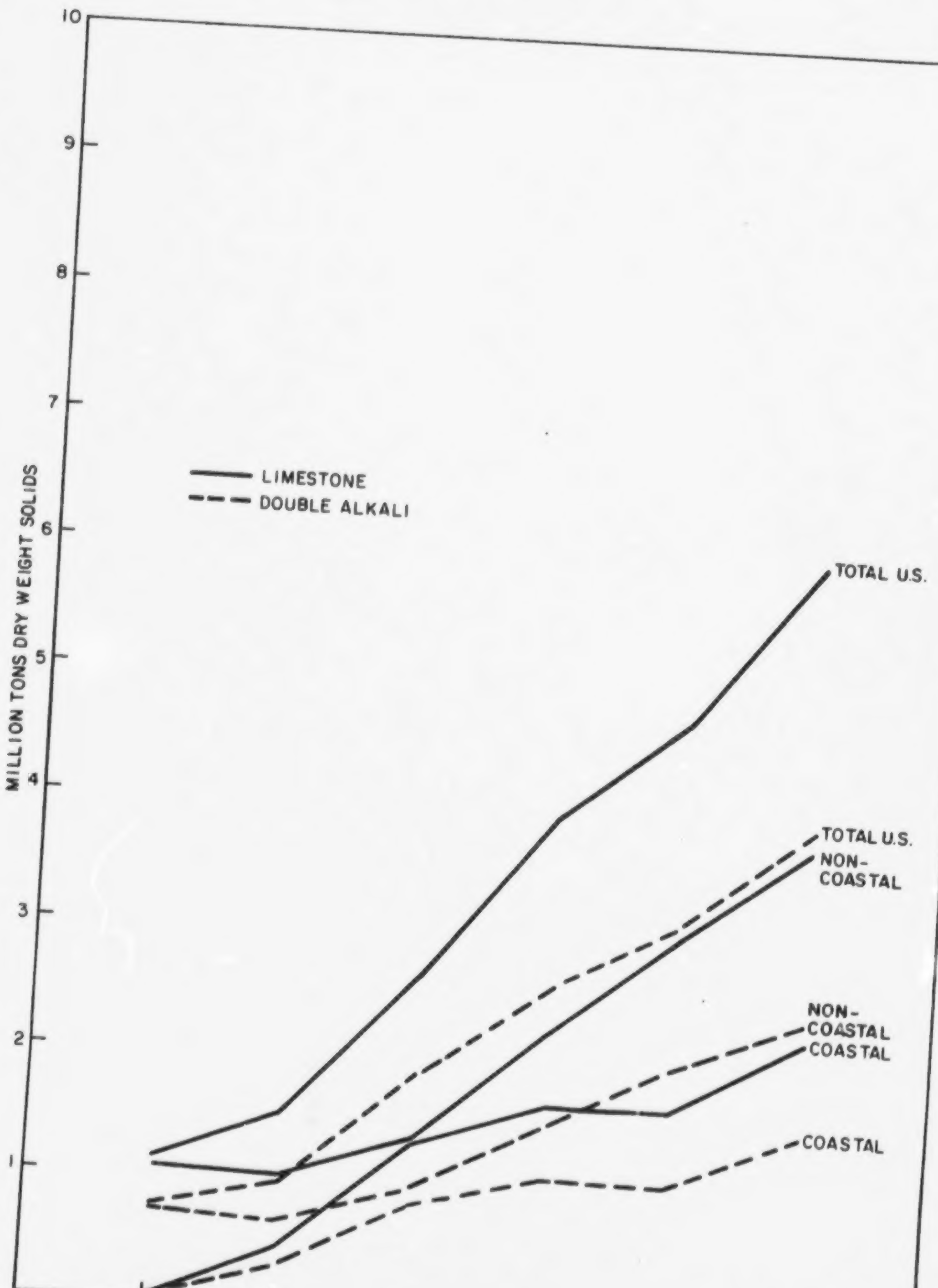


Figure 2-4. Quantity of flue gas desulfurization sludge produced by electric utilities in the United States 1975-1980.

FGD sludge quantities (Figure 2-4) present a more complex situation, with quantities generated being dependent on a wider variety of factors including the sulfur content of the coal, the age and load factor of particular power plants, and the FGD technology utilized (lime, limestone, or double alkali). Before 1975, very few power plants employed FGD systems. The introduction of more stringent air emissions limitations and New Source Performance Standards have and will continue to result in increasing levels of sulfur recovery through the installation of FGD systems. Between 1975 and 1980, the capacity of electric power plants with FGD systems has increased more than 600% from 3.86 million megawatts in 1975 to 27.92 megawatts in 1980 (DOE, 1981). The quantities of FGD sludge generated have risen similarly over this period by roughly 430%. In 1975, just over 1.1 million tons of FGD sludge were produced, which compares to an estimated 6 million tons in 1980. (These estimates assume the utilization of limestone systems. The use of double alkali technology would result in estimates of 760,000 tons and 3.9 million tons in 1975 and 1980, respectively.) An examination of the total quantity of sulfur dioxide (SO_2) produced by electric utilities helps to put these estimates in perspective. Based on the quantities and sulfur content of coal consumed by utilities, an estimated 15.2 million tons of SO_2 were produced in 1980 and 12.6 million tons in 1975 (an increase of 21%). It can be inferred, therefore, that in 1980, less than 11% of the SO_2 produced was captured in the form of FGD sludge. (Approximately 543 pounds of SO_2 are incorporated into each ton of limestone FGD sludge produced.) Clearly, the quantities of FGD sludge which could be generated in the future are dependent on factors other than the quantity of coal likely to be consumed. The sulfur content of this coal, along with the trends in FGD system installation and technological advances, will also be influential factors.

According to the U.S. Bureau of Mines (1975, 1976), nearly 55% of U.S. coal reserves are located in the western half of the country, and nearly 80% of these reserves have a sulfur content under 1%. Roughly 80% of eastern coals have a sulfur content which exceeds 1%. Western coals are also characterized by higher ash content and lower heat value, and are more readily available using surface recovery techniques. OTA (1979) projections indicate that coal production from western mines will increase in importance through the year 2000. In 1977, only 22% of domestically produced coal came from western states while the OTA projections call for the proportion to increase to more than 50%.

Although there is some controversy over the effectiveness, reliability, and economics of FGD systems, their use is expected to continue to increase in the future. The U.S. Environmental Protection Agency (EPA, 1979) estimates that the quantity of electricity generated by plants with FGD systems will rise from 2.5 billion kwh to 500 billion kwh by 1985. Since the EPA report was published, FGD capacity has increased by less than 50%, making such a radical increase in FGD capacity by 1985 difficult to achieve at best.

With regard to technological improvements, SO_2 removal efficiencies of 90 to 95% appear to be achievable (OTA, 1979). The major problem with FGD systems has been one of reliability, and advances in this aspect of operation are likely. Recent experience in Japan has shown that 95% operating reliability and 90% sulfur removal efficiencies can be regularly achieved (Maxwell, 1978).

In summary, it is apparent that the quantities of both ash and FGD sludge generated by electric utilities are likely to increase in the coming years. Ash quantities will rise as coal grows in importance as a utility fuel, and as western coals become a more important component in the overall use of coal. The situation with FGD sludge is more complex, but an increasing trend in its generation is likely. Estimates presented by the EPA (1979) and OTA (1979) support this observation. Table 2-3 summarizes these and other estimates of future ash and FGD sludge quantities.

TABLE 2-3. PROJECTIONS OF ASH AND FGD SLUDGE PRODUCTION FOR ELECTRIC UTILITIES BURNING COAL.
(10^6 TONS)

Source	Ash		FGD Sludge	
	1985	1990	1985	1990
EPA (1979)	108	---	28	--
OTA (1979)	80	90	19	35
U.S. Bureau of Mines (1982)	100	112	--	--
National Ash Association (1982)	90	125	--	--
Mean	94.5	109	23.5	35

The means of these estimates for ash in 1985 and 1990 represent increases of 40% and 60%, respectively, from 1981 levels. The FGD sludge estimates indicate increases of 400% and 580%, over 1980 levels.

2.1.4 Gypsum from the Basic Fertilizer Materials Industry

Gypsum is an important by-product of fertilizer production. The manufacture of ortho-phosphoric acid, ammonium phosphate, and concentrated superphosphate represents a key component in the U.S. fertilizer industry. Consequently, the generation of waste gypsum from the fertilizer industry is closely related to both U.S. and worldwide fertilizer usage, which is dependent, in turn, upon the level of agricultural activity and cropping patterns. From 1954 through 1970, fertilizer consumption in the U.S. increased 83% from 22.3 to 40.8 million tons (EPA, 1974 and CEQ, 1981). Worldwide consumption of fertilizer increased by more than 100% during the same period (CEQ, 1981).

The production capacity of U.S. plants producing phosphate-based fertilizer materials has increased steadily since 1976. Plant capacity figures published by the Tennessee Valley Authority (1979) are summarized in Table 2-4, and indicate a 3.6% increase in capacity between 1976 and 1979. The projected total capacity in 1983 is 19.2 million tons, which represents an increase of 14% over 1979 levels. Waste quantities of gypsum resulting from production at capacity levels were derived by applying waste coefficients presented in EPA (1977). These estimates and projections through 1983 are shown in Figure 2-5. Production at levels less than capacity would results in proportional decreases in the generation of gypsum waste.

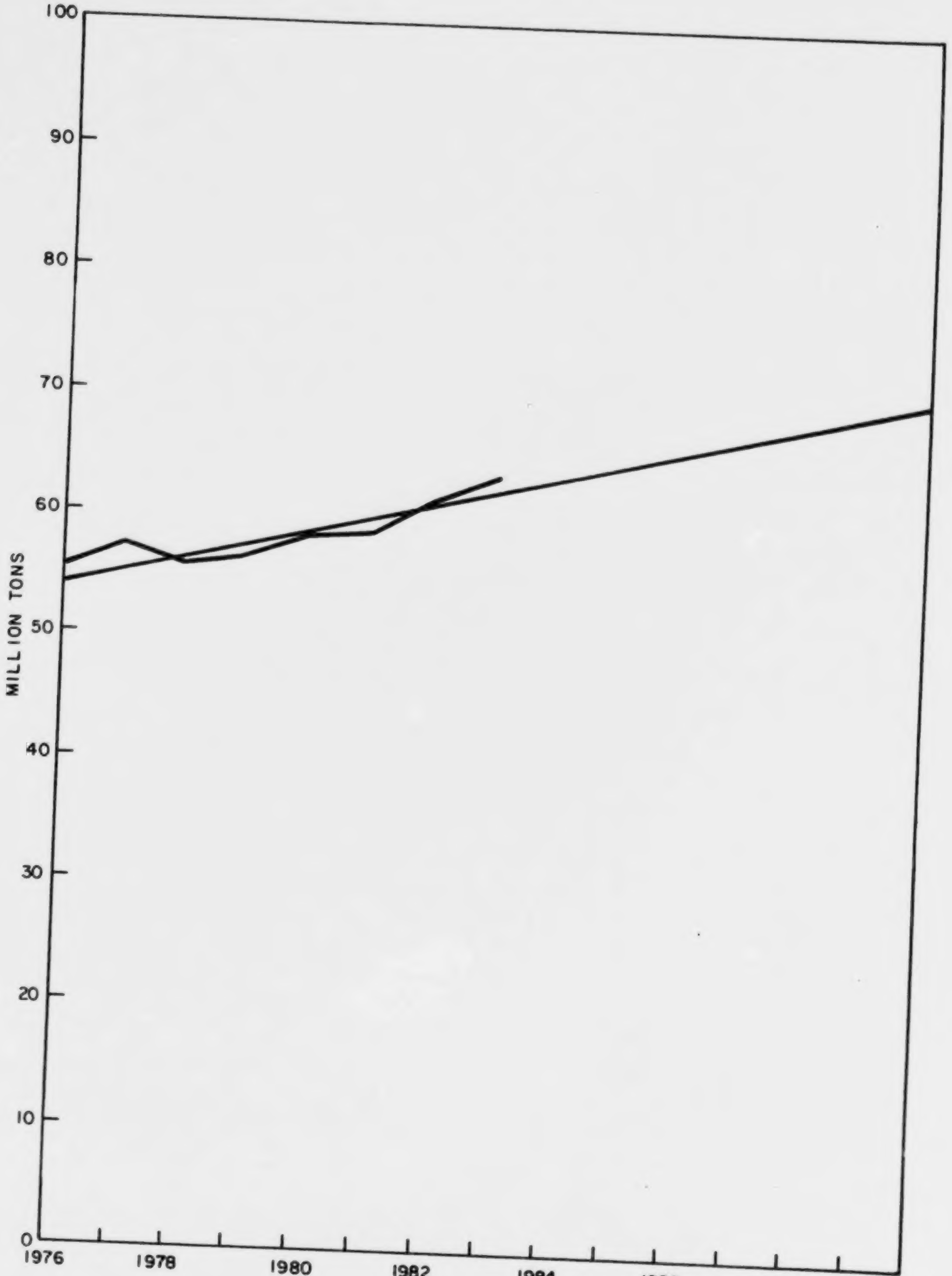


Figure 2-5. Quantity of gypsum produced as a by-product in the basic fertilizer industry 1976-1981 with projections.

TABLE 2-4. BASIC FERTILIZER MATERIALS INDUSTRY PRODUCTION CAPACITY PHOSPHATE-BASED MATERIALS. (1000 TONS)

	1976	1977	1978	1979	% Increase
Wet-Process Phosphoric Acid	8,951	9,296	9,561	9,689	8.2
Ammonium Phosphate	4,721	5,155	4,682	4,725	0.1
Concentrated Superphosphate	2,596	2,440	2,440	2,440	-6.0
Total	16,268	16,891	16,683	16,854	3.6

2.1.5 Waste Acid Production in the Titanium Dioxide Industry

The mining and concentrating of titanium and the production of titanium dioxide results in the generation of large quantities of wastes. As noted by the EPA (1977), the disposal of these wastes presents a major environmental problem to the industry. The major waste of concern consists of sulfuric acid that has been either neutralized or ocean dumped.

The most common processes for producing titanium dioxide are the sulfate and chloride processes. According to the EPA (1977), each metric ton of titanium dioxide produced by the sulfate process results in the generation of 2.4 metric tons of sulfuric acid, 90% of which is "weak" acid (1.35% H_2SO_2) and 10% of which is "strong" acid (18% H_2SO_2). The chloride process produces lesser volumes of waste acids (typically 0.6 metric ton/metric ton output), with roughly the same acid content.

During 1972, the production capacity of the titanium dioxide industry in the U.S. was 852,300 metric tons, of which 458,400 metric tons were based on the chloride process and 393,900 metric tons on the sulfate process. During 1972, the industry operated at approximately 82.5% of capacity for an implied output of 703,148 metric tons of titanium dioxide. In 1973 and 1974, the industry produced approximately 650,000 and 715,000 metric tons of product, respectively (May and McManus, 1977 and EPA, 1977). According to Trees et al. (1979), 15% of the acid in the raw waste stream can be recovered and reused. Assuming that industrial capacity remained constant during the 1972-1974 period, the quantity of waste acid can be estimated by combining the waste acid coefficients and the titanium dioxide production estimates. These quantities are presented in Table 2-5.

As shown, an estimated annual average of 838,882 tons of waste acid was produced from 1972 to 1974. Since the actual waste stream contains a number of other compounds (including ferrous sulfate and other assorted sulfates and metal salts), the quantities in need of disposal will be somewhat higher. For example, an estimated 1.7 million metric tons of iron-acid sludge and 146,000 metric tons of iron-chloride sludge were produced in 1972 (EPA, 1977).

TABLE 2-5. WASTE ACID PRODUCTION IN THE TITANIUM DIOXIDE INDUSTRY 1972-1974. (METRIC TONS)

	1972	1973	1974
Titanium Dioxide Production			
Chloride Process	378,294	349,700	384,670
Sulfate Process	324,854	300,300	330,330
Total	703,148	650,000	715,000
Waste Acid Production			
Weak Acid	905,963	837,486	921,235
Strong Acid	100,663	93,054	102,359
Total	1,006,626	930,540	1,023,594
Reprocessed Acid	150,994	139,581	153,539
Net Waste Acid	855,632	790,959	870,055

Acid wastes dumped in the New York Bight by NL Industries contain approximately 8.5% sulfuric acid (Anderson et al., 1979). If this is representative of "typical" acid waste from titanium dioxide production, the estimates in Table 2-5 can be adjusted to figures which represent the total quantity of waste material in need of disposal. For 1972 to 1974 these quantities would have been 10.1, 9.3, and 10.2 million metric tons, respectively.

Titanium dioxide is used as a pigment in paint, paper, plastics, floor coverings, synthetic fibers, and rubber manufacturing. Consequently, the quantities of acid wastes generated by the production of titanium dioxide are closely related to trends in these associated industries. There is, at present, no reason to believe that these industries' use of titanium dioxide will decline in coming years.

2.1.6 Low-Level Radioactive Waste (LLW)

Low-level radioactive wastes result from the generation of electricity in nuclear power plants, government and industrial research, medical isotope production and use, and industrial manufacturing. In 1978, commercial nuclear power plants accounted for 43% of the volume of LLW generated in the U.S. and disposed of at commercial disposal sites (NUS, 1980a). During 1979, the electric utility share of LLW production increased to nearly 50% (NUS, 1980b). Institutional and industrial sources accounted for 50% and 41% during 1978 and 1979, respectively. Figure 2-6 illustrates the volume of commercially (non-government) produced LLW annually from 1962 through 1979.

The vast majority of LLW produced by government facilities is disposed of at government-operated sites and is not considered in these figures. More than two-thirds of the LLW disposed of at U.S. burial sites through 1975 was of government origin. Obviously, LLW activity at government facilities is important, particularly when considering the volumes of waste in need of disposal in future years.

Figure 2-6 also presents two assessments of such future quantities. The DOE (1980) estimate is for commercially produced LLW only, while that of the California Energy Commission (CEC, 1979) considers both commercial and government sources. The figures for 1990 reflect past LLW disposal experience with commercial wastes estimated at roughly 264,000 cubic meters, compared to the 1-1.2 million cubic meter estimate for the combined commercial/government waste volume. The CEC estimates assume fairly significant growth in the nuclear power industry through 1990, with nuclear generation capacity increasing 128-150%. By 1990, they speculate that nuclear power plants will generate 58-63% of the U.S. total LLW.

Given recent emphasis on increased utilization of coal for electricity generation and the diminishing emphasis on continued nuclear power growth, the CEC projections are not likely to be realized. A volume estimate of 500,000 cubic meters of LLW in 1990 is probably more reasonable. Volume reduction technologies may further modify these future volumes.

2.1.7 Seafood Processing Waste

Seafood wastes are generated by firms that process and package finfish and shellfish. The volumes of waste generated are a function of the type of fish processed, the degree of processing, and the quantity of fish involved. The seafood waste quantities presented in Figure 2-7 for the 1970-1980 period were derived by applying waste quantity coefficients to particular types of fish landed each year. Fish landing data were obtained from various National Marine Fisheries Service statistical reports (NMFS, 1973-1981). Waste quantity data for specific fish types were obtained from a variety of studies including Murray (1981), Perry (1981), Brown (1981), NMFS (1973), Pigott (1981), and Webb et al. (1976).

Between 1970 and 1980, the weight of landed fish meat increased 33% (2442 thousand tons to 3241 thousand tons). Landings of clams, oysters, shrimp, and lobsters declined by an aggregate total of 6%, while scallop, crab, and finfish landings increased by 38%. Over the same 10-year period, seafood wastes increased in quantity by an estimated 35-41%. The range in waste output shown in Figure 2-7 stems from the possible ranges in waste composition associated with particular types of fish and processing techniques. During 1980, approximately 1.4-2.0 million tons of seafood waste were produced in the U.S. This compares to the estimated 0.9-1.5 million tons generated in 1970. The vast majority of these wastes (35-51%) resulted from the processing of finfish.

Future waste quantities will be influenced by several factors, including the quantities of fish landed and processed, the composition of the landings (shellfish versus finfish), and the degree to which waste products are utlized for animal feed and fertilizer.

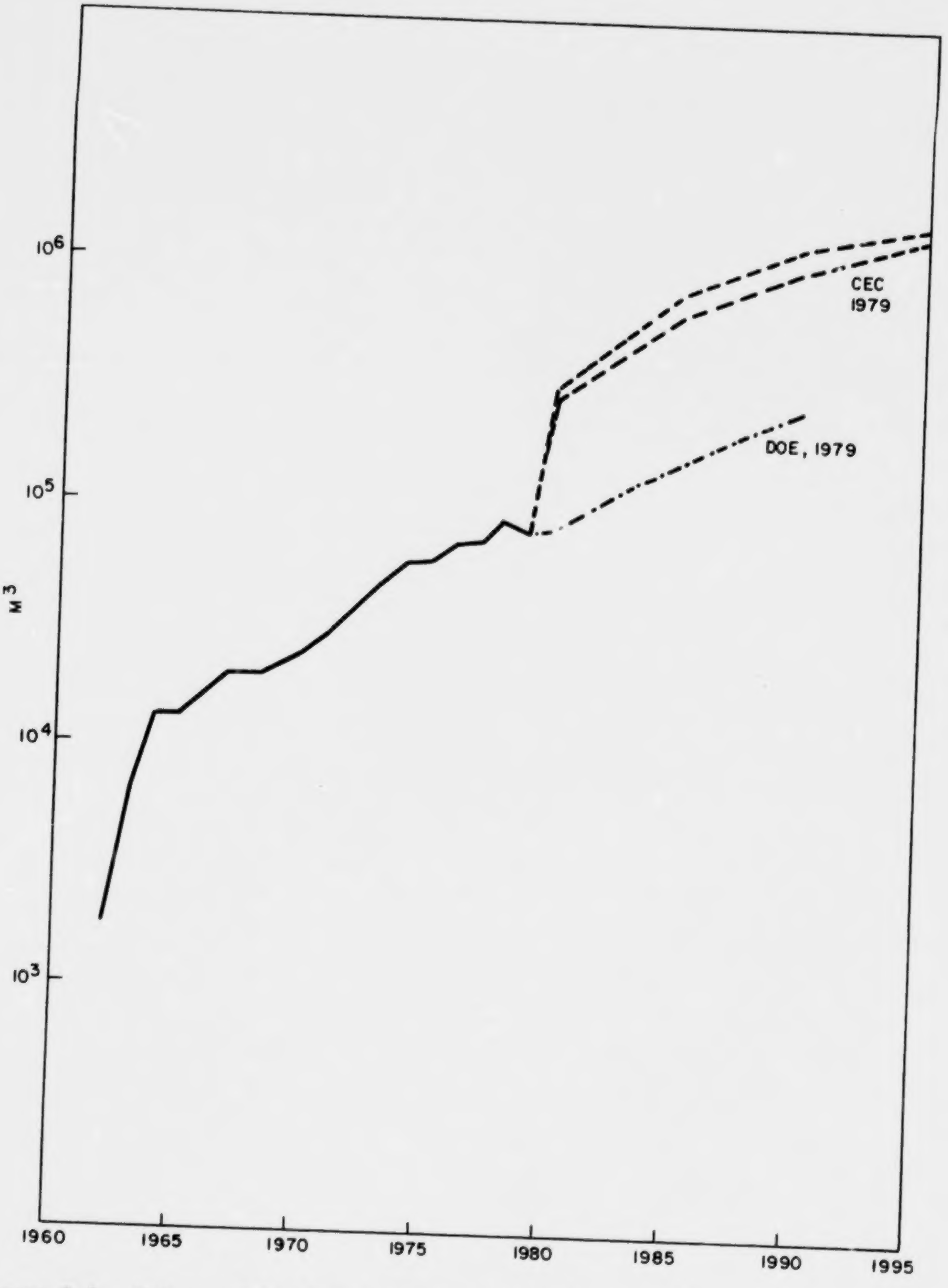


Figure 2-6. Volume of low-level radioactive waste produced and disposed of at commercial facilities in the United States 1962-1979 with projections.

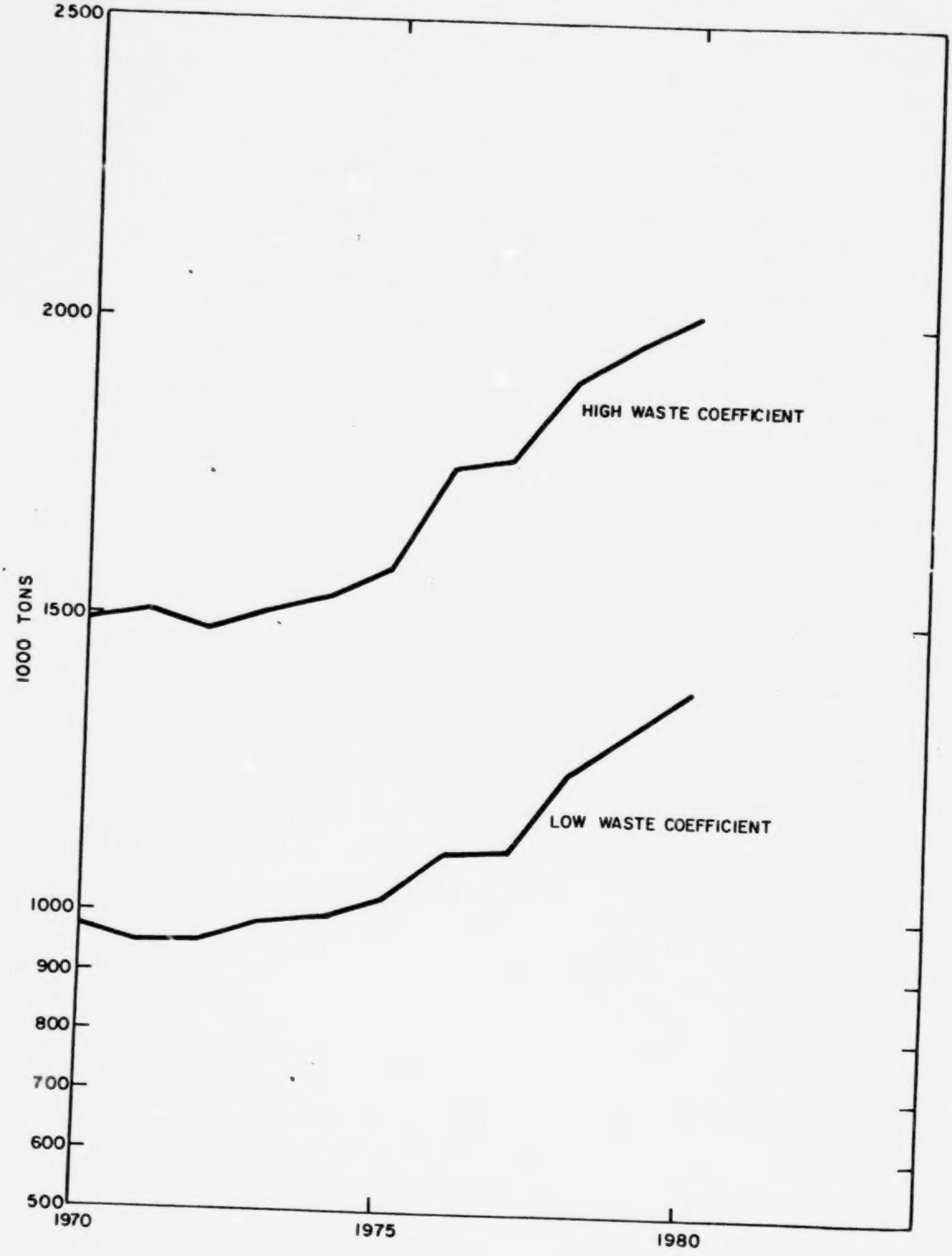


Figure 2-7. Seafood waste quantities.

2.2 CURRENT DISTRIBUTION OF WASTES AMONG DISPOSAL OPTIONS

2.2.1 Sewage Sludge

An estimated 6.5 million dry metric tons of sewage sludge were generated in 1980. The primary disposal methods for sewage sludge are landfilling, incineration, land application, wet ponding, and ocean dumping. The quantities of sewage sludge disposed of by these various options have been estimated by several investigators (Farrel, 1975; Champ and Park, 1981; Hyde, 1981; and U.S. GAO, 1978) and show considerable variation. Table 2-6 summarizes this information.

TABLE 2-6. SEWAGE SLUDGE DISPOSAL METHODS AND RELATIVE USE.

	%
Landfill	15-33
Land Application	25
Wet Ponding	0-11
Incineration	22-35
Ocean Dumping	4-15

Landfilling is the major disposal option for sewage sludge when one considers that the ash resulting from incineration is almost exclusively landfilled. Land-based disposal (including landfilling, land application, ponding, and incineration) accounts for 85-96% of the disposed sludge. This translates to roughly 6 million dry metric tons annually. The remaining tonnage is ocean disposed by either dumping or pipeline discharge.

2.2.2 Dredged Material

The ultimate disposal options employed for dredged materials are difficult to identify. Data are collected only for the ocean dumping option and "other methods" (e.g., beach replenishment, upland disposal) are not accounted for in terms of quantities. In 1981, an estimated 22% (74 million cubic yards) of the 335 million cubic yards of dredged material were ocean dumped. Between 1974 and 1980, an annual average of 21.6% (66.7 million cubic yards) of dredged material was ocean dumped.

2.2.3 Coal Ash from Coal Use by Electric Utilities

Coal ash, produced as fly ash, bottom ash, and boiler slag, has found increasing use in commercial applications. In 1981, the production of fly ash, bottom ash, and slag totalled 48.6, 14.1, and 5.2 million tons, respectively. Table 2-7 summarizes the relative utilization of various disposal options for these three types of coal ash.

TABLE 2-7. COAL ASH DISPOSAL OPTIONS AND THEIR RELATIVE USE.
(MILLION TONS [%])

	Fly Ash	Bottom Ash	Boiler Slag	Total
Commercial Use	6.2 (13)	4.7 (33)	3.1 (60)	14.1 (21)
Landfill	21.4 (44)	4.8 (34)	1.0 (20)	27.2 (40)
Wet Ponding	20.9 (43)	4.7 (33)	1.0 (20)	26.6 (39)

Landfilling and wet ponding represent the key disposal options for coal ash. The choice between the two depends primarily on the moisture content of the ash, which depends in turn on the combustion and ash recovery technologies utilized at particular power plants. Dry disposal in landfills appears to be the preferred future option, due in part to regulatory constraints at local and federal levels. Commercial uses for coal ash are expected to grow in importance in the future, particularly in more applications related to cement manufacture, road bedding, sand blasting, roofing, and snow and ice control.

2.2.4 Flue Gas Desulfurization (FGD) Sludge

More than half of the FGD sludge generated in the U.S. during 1980 was disposed of in either lined or unlined ponds. Unlined ponds have been the dominant method, receiving between 2 and 3 million dry tons in 1980. Roughly 30% of the FGD sludge produced was disposed of in landfills, and about half of this volume was chemically treated prior to landfilling. A small quantity of sludge (2%) was disposed of in mines.

Growing concerns with groundwater protection and land reclamation are likely to lead to increased emphasis on secure landfills and lined ponds as FGD sludge disposal options.

2.2.5 Gypsum from the Basic Fertilizer Materials Industry

Data describing volumes and methods for the disposal of waste gypsum are sparse. In general, gypsum by-products are stored on-site by wet ponding, or if the wastes are dewatered, by stockpiling. An estimated 10% of the total volume of gypsum produced in the fertilizer industry is utilized commercially in construction-related activities. There is no evidence that commercial users of gypsum will be able to expand their consumption of this material.

2.2.6 Waste Acid from the Titanium Dioxide Industry

Approximately 25% of the by-product acid resulting from titanium dioxide production has been used commercially in recent years. The purification process is costly and places this source of industrial acid in a relatively weak competitive position. Other disposal options for the waste acid include chemical neutralization (with lime or limestone), dilution and discharge, and ocean dumping. Neutralization accounts for approximately 25% of the waste acid produced. The resulting by-product is gypsum, which, as described for fertilizer producers, must be stored on-site due to unfavorable market conditions for commercial gypsum suppliers. An estimated 20% of the waste acid is discharged to temporary on-site storage, diluted, and discharged; or diluted and discharged without temporary storage. The remaining 30% of produced waste acid is ocean dumped, primarily on the northeast continental shelf.

Other available options exist for waste acid disposal, including submerged combustion to produce sulfur oxides for iron oxide and/or sulfuric acid production, and neutralization with ammonia, caustic or soda ash, or magnesium hydroxide. The former was judged extremely costly in terms of both energy use and maintenance requirements by Trees et al. (1979). The other neutralization processes, which lead to marketable products, were thought to be of limited value because the resulting products' markets are saturated. It appears as though neutralization with lime or limestone will be a key future disposal option, leading to the generation of waste sludges and gypsum.

2.2.7 Low-Level Radioactive Waste (LLW)

At the present time, LLW is disposed of exclusively by landfilling. since 1978 there have been only three sites available for LLW disposal, in South Carolina, Nevada, and Washington. Of the 80,000 cubic meters of LLW disposed of at commercial disposal sites in 1979, 79% went to the South Carolina site, accounting for 66% of the radioactivity contained in the LLW. The key states in terms of volume produced in 1979 were New York (12%), South Carolina (10%), Illinois (8%), Pennsylvania (9%), North Carolina (7%), and Massachusetts (6%). In terms of radioactivity content, Massachsuetts (29%), Florida (19%), New York (17%), and California (17%) represent the major producers.

A study completed by the California Energy Commission in 1979 (CEC, 1979) estimates that the three operating sites could be filled by 1990. The study identifies shallow land burial and on-site storage and disposal as the two most viable disposal options in the future. Ocean dumping of LLW, practiced by the U.S. between 1946 and 1970, was cited by the Nuclear Regulatory Commission (USNRC, 1978) as an option that is technically feasible but politically difficult.

2.2.8 Seafood Processing Waste

Several alternative disposal methods have been used for seafood processing wastes, including landfilling, land application, grinding/drying for meal, NPDES-controlled discharging, and ocean dumping. Unfortunately, data do not exist which describe the distribution of waste quantities among the available disposal options.

2.2.9 Industrial Hazardous Wastes

A survey conducted as part of this study revealed that approximately 84% of the industrially produced hazardous waste was recycled or disposed of at industrial sites via methods other than offsite disposal. The remaining 16% were disposed offsite by landfilling (4.9%), ocean dumping (4.0%), chemical treatment (3.5%), deep-well injection (1.3%), land application (0.9%), resource recovery (0.7%), and incineration (0.7%). Specific "on-site" disposal methods were not identified. Regulatory constraints are likely to cause landfill utilization to decline in the future in favor of incineration, resource recovery, deep-well injection, and other methods.

2.3 SURVEY OF U.S. EPA REGIONS

In an effort to assess likely trends in ocean dumping practices that may result from promulgation of less stringent ocean dumping guidelines, a survey was conducted of all U.S. EPA Regional Ocean Dumping Coordinators. Officials at EPA coastal regions I, II, III, IV, VI, IX, and X (see Figure 2-8) were surveyed to ascertain what they perceived will be the demand for ocean dumping in the future. The results of this survey are summarized in Table 2-8.

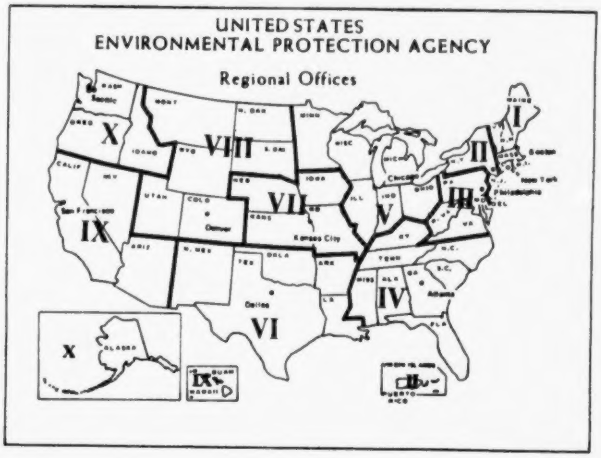


Figure 2-8. Map of U.S. EPA standard federal regions and regional offices.

The survey results indicate likely trends for ocean dumping, based on the historical perspective of the regional coordinators together with their views of recent events in their regions. The significance of these projections should be viewed in light of the history of ocean dumping in each region, as well as current and projected industrial and population growth of major economic centers within each region. The following is a short summary, organized by waste type, of expected ocean dumping trends or demands on using this alternative as expressed by the EPA regional coordinators.

2.3.1 Sewage Sludge

The thrust of the 1977 ocean dumping regulations has been to phase out sewage sludge dumping. The current version of 301(h) regulations also prohibits ocean disposal of sewage sludge. The Region II coordinator believes that this position will prevail and ocean disposal of sewage sludge will continue to be prohibited. However, officials interviewed at all the other regions, except Region IV, believe that municipalities will be able to demonstrate that no reasonable alternatives exist and that the dumping will not unreasonably degrade

TABLE 2-8. RESULTS OF TELEPHONE SURVEY OF U.S. EPA REGIONAL OCEAN DUMPING COORDINATORS

WASTE	USEPA REGIONS							REMARKS
	1	2	3	4	6	9	10	
Sewage Sludge	I	D*	I	I	I	I	I	*Region 2 believes 1977 regulations will prevail and continue to prohibit this disposal.
Dredged Material	S	S	S	S	I[1]	I[2]	I[2]	[1]S.W. Ind. Growth. [2]Deepening Projects.
Fly Ash	I	I	S	S	NR	S	S	Region 10 has very little now.
RCRA Driven Chemical Waste	I	I	I	I	I	I	I	Also avoiding pretreatment requirements.
Gypsum Sludge	NR	I	I	NR	I	S	NR	Gypsum sludge parallels the increase in chemical waste.
Low-Level Rad Waste	I	I		I	I			
Ocean Incineration	I	I	NR	I	I	NR	NR	Ships being built for operation in any region.
Construction/Demolition Debris	I	S	S	NR	NR	NR	I*	*Primarily from Alaska.
Cannery Waste	NR	NR	NR	NR	NR	I	NR	Only reported in Region 9.
NSSC Liquor	NR	NR	NR	NR	NR	NR	I	Only reported in Region 10.
Secondary Sludge from Pulp and Paper	NR	NR	NR	NR	NR	NR	I	Only from Region 10.
Phosphate Mining	NR	NR	NR	I	NR	NR	NR	Reported only in Region 4, uncertain whether O.D. permit will be required.
Drilling Muds	NR	NR	NR	I	I	I	NR	Likely only where nearby deep water is available.
Deep Ocean Mining	NR	NR	NR	NR	NR	I*	NR	*May or may not be an Ocean Dumping permit.
Tires	I	I	I	D	NR	NR	NR	Principally as proposals to construct fish reefs.
Vessels	S	S	S	S	NR	NR	S	Vessels will continue to be scuttled for various reasons.

I = Increase
D = Decrease
S = Same
NR = Not Reported

the marine environment. Some municipalities such as the South Essex Sewage District, Massachusetts, may find it economically attractive to ocean dump rather than incinerate or build a long outfall pipe to meet a 301(h) demonstration.

2.3.2 Dredged Material

All coastal EPA regions are involved in the issuance of ocean dumping permits for dredged material by the U.S. Army Corps of Engineers. The Corps' own dredging projects must also comply with ocean dumping criteria if dredged material is disposed of in the "ocean" (seaward of the baseline). Most of the regions believe that the demand for dredging (maintenance dredging of federal projects) will continue, dependent on sedimentation rates and the availability of federal funds. Increases in the ocean disposal of dredged material are expected in Region VI as industrial growth in the southwest continues, and in Regions IX and X due to deepening projects to accommodate larger draft vessels.

2.2.3 Fly Ash

Officials at Regions I and II believe that there will be an increased demand for ocean dumping of fly ash. The use of coal for generating electricity is expected to increase in these regions (see subsection 2.1) and there will be more fly ash produced than can reasonably be landfilled. Other coastal regions do not anticipate any demand for ocean dump ash; in the west and northwest the coal plants are relatively far inland, resulting in economic disincentives for the ocean disposal option.

2.3.4 RCRA-Driven Chemical Waste and Pretreatment Requirements

All the coastal regions are convinced that there will be increased pressure for ocean dumping due to regulatory control of solid wastes under the Resource Conservation and Recovery Act (RCRA) and requirements for pretreatment of industrial waste. Many of these wastes, produced by industrial processes and considered hazardous by the EPA, would be prohibited from ocean dumping by the London Dumping Convention. Ocean disposal of these wastes may be less expensive for waste generators in the long run because of the high cost of long-term monitoring and land-disposal sites. Another possible candidate for ocean dumping is stabilized hazardous waste. This material can be rendered nonhazardous by chemical processing and dumped at sea, thus avoiding the costly siting, closure, and monitoring requirements of a landfill.

2.3.5 Gypsum

Although not reported in Regions I, IV, and X, production of gypsum sludge (from the basic fertilizer materials industry and the neutralization of waste industrial acids) and the need to ocean dump this material is likely to increase and follow the trend of increased amounts of chemical waste.

2.3.6 Low-Level Radioactive Waste (LLW)

Most regions believe that there will be an increased demand for dumping low-level radioactive waste at sea.

2.3.7 Ocean Incineration

Region VI currently has active permits for high-temperature incineration at sea. The materials being incinerated (e.g., agent orange) are loaded at Mobile, Alabama in Region IV. This technique is capable of thermally destroying waste pesticides, herbicides, and other chlorinated organic compounds at greater than 99% efficiency. At-Sea Incineration is building two ships for this purpose, and Ocean Combustion is building a ship similar to the original VULCANUS. The Region I and II coordinators also believe that their regions have potential locations for ocean incineration.

2.3.8 Construction/Demolition Debris

Regions I, II, and III have traditionally permitted dumping of inert materials resulting from the demolition of buildings and other structures. In Region I, the amount dumped is expected to increase as landfills become increasingly expensive to use. Regions II and III have always dumped significant amounts of debris and, although the quantity generally has been decreasing, dumping is expected to continue and may show increases. Region X expects an increase in debris dumping in the near term, primarily from military bases in Alaska.

2.3.9 Seafood Processing Waste

Only Region IX reported that it may receive requests to dump seafood processing wastes, primarily the wastewater treatment sludge from tuna canneries. This waste category does not include unprocessed seafood waste (from cleaning fish prior to processing), which is excluded from regulation by 40 CFR 220.1(c)(1).

2.3.10 Waste Liquor

Waste liquor from paper pulping may have to be dumped if markets cannot absorb the amount produced. Paper plants located in Washington state may find it less costly to barge this material offshore for dumping than to treat and discharge it.

2.3.11 Secondary Sludge from Pulp and Paper

The Region X coordinator reported that there will be a demand to dump secondary sludge from pulp and paper waste treatment. Mills located in the Puget Sound area may find it economically advantageous to transport this sludge beyond the nearby continental shelf edge for dumping.

2.3.12 Phosphate Mining

Region IV reported that phosphate mining contemplated for Pamlico County, North Carolina, and in shallow water off Georgia may generate waste mining and beneficiation material that could potentially be ocean dumped. This is the only region in the country that reported this potential activity, and the likelihood of it occurring or the quantities of material involved are not known at this time.

2.3.13 Drilling Muds

In some locations, direct discharges of drilling muds offshore is prohibited by regulatory agencies due to the ecological sensitivity of the area surrounding the drill site. Regions IV, VI, and IX have required that muds be barged and dumped away from drilling sites. As offshore activity increases and new lease areas are explored, it is expected that the number of applications for ocean dumping permits for this material will increase.

2.3.14 Deep Ocean Mining

Region IX may be requested to issue ocean dumping permits for deep ocean mining activities. The requirement for ocean dumping permits for this type of operation will depend on the type of discharge (bulk discharge from a vessel or continuous point source) and whether the mining equipment will be considered a "vessel or other floating craft" (33 USC 466 Sec. 502).

2.3.15 Tires

Rubber tires have been disposed of offshore by creating "tire reefs" out of thousands of tires which are bound together and sunk. To date, no land-based disposal method has been developed to successfully deal with the huge quantity of discarded tires generated each year. As this material builds up at various land locations, regions in the northeast believe they will be presented with more proposals to create "reefs."

A major reason for disposing of rubber tires as reefs is that under 40 CFR 220.1(c)(2) the placement of materials for fishery enhancement is exempt from regulation under the Ocean Dumping Act, provided that the placement of these materials is part of an authorized state or federal fisheries enhancement program. Also the placement of tires as a reef unit helps to ensure that they will remain where they are placed.

2.3.16 Vessels

Vessels are often towed out to sea and sunk because this is the least expensive disposal method. Regulation 40 CFR 229.3 provides for a general permit to allow this disposal, provided that all material that may pollute the marine environment is removed from the vessel and provided that the dump site is at least 12 miles from the nearest land and at least 300 feet deep. The need for this type of ocean disposal is expected to continue.

2.4 SURVEY OF MUNICIPALITIES

Representatives of the Conference on Coastal Agencies (CCA), a subcommittee of the Association of Metropolitan Sewerage Agencies (AMSA), were surveyed to determine current and future interest in ocean dumping for their districts' sewage sludge disposal needs. We contacted 14 urban coastal sewage districts, from Alaska to Maryland, with populations ranging from 175,000 (Anchorage, Alaska) to 8 million (New York City). Although this was not a statistical survey, the results do highlight issues critical to the ocean dumping decision process.

In general, municipalities only appeared interested in ocean dumping when land-based alternatives were unavailable, expensive, or prohibited. Otherwise, the capital and operational costs of sludge barging, the time and expense associated with dump site selection and approval, and public opinion make the ocean dumping option less desirable.

Most of the sewage districts surveyed foresaw at least a minor increase in sludge volume from 1) population increase, 2) enlargement of sewage districts (e.g., Orange County Sanitation Districts, California), 3) start-up of secondary treatment plants (e.g., Passaic Valley Sewerage Commission, Newark, New Jersey), and 4) increase in the proportion of the population sewered (e.g., Nassau County, New York). However, prediction is often difficult; for example, the Director of Public Works, City of Baltimore, Maryland, said future trends in sewage volume in his district depend on the the housing market and population shifts between the city and suburbs. The South Essex Sewage District, Massachusetts, actually may experience a volume decrease as industry (leather tanning) moves out of the area.

Representatives of several districts remarked on legislative guidelines that make land-based sludge utilization options impractical and may increase the desirability of ocean dumping. For example, the Public Works Utility Services, Tacoma, Washington, has had a sludge utilization plan (fertilization) for 30 years. However, EPA guidelines specifying secondary treatment and trace metal levels are limiting the continuation of this land-based alternative.

2.5 SURVEY OF INDUSTRIES

In addition to assessing the ocean dumping prospects of the major waste categories considered above, we also surveyed representatives of several other waste-generating industries that have ocean dumped in the past. The intent was to assess the potential future attractiveness of ocean dumping to industries that have already developed land-based disposal. The three major industry groups surveyed were pulp and paper, chemical, and pharmaceutical companies.

2.5.1 Pulp and Paper

According to the National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI), the main use of the ocean by the paper industry has occurred in the Pacific Northwest where proximity to deep water made ocean disposal of process effluents attractive, economically and environmentally. In addition, some paper mills have used municipalities (~10% of the industry) or private contractors to barge solid and liquid wastes to sea. In these cases the barging was cost effective relative to concentrating and disposing of these wastes on land (James J. McKeown, NCASI, pers. comm.).

Although the paper industry has not extensively used the ocean to dispose of its wastes, there are cases where the ocean offers an economical and feasible alternative for disposal. Therefore, NCASI is interested in seeing research supported that will determine the impact and fate of various paper industry waste materials if ocean disposed. Large pulp and paper operations are able to burn much of their wastes for heat recovery, product recovery, or waste volume reduction. However, for small companies, the cost of a furnace is high and a small-scale furnace has poor heat balance.

One factor that has reduced industry interest in ocean dumping is that many existing companies have already installed Best Practical Technology (BPT). A major determinant of the attractiveness of the ocean dumping option, therefore, will be which guidelines are in effect for new facilities.

2.5.2 Chemical

According to a representative of a major chemical company, it appears unlikely that wastes previously disposed of in the ocean and now land disposed would again be ocean dumped if this option were available. Most large companies have made substantial investments in incinerators, deep wells, and other land-based facilities to dispose of their wastes. Future applications for ocean disposal would depend on regulatory requirements and the results of evaluating alternatives for the disposal of specific wastes (R.F. Schwer, E.I. duPont de Nemours & Company, pers. comm.). A representative of another major chemical company discussed the possibility that industrial wastes similar to those presently being ocean dumped (e.g., waste acids) would continue to be dumped and would perhaps increase in the future.

Other industry representatives surveyed specified heavy metal disposal as a major category of concern, because of difficulty in heavy metal "detoxification." They expressed particular interest in research on the fate of heavy metals in the ocean.

2.5.3 Pharmaceutical

Most pharmaceutical companies now have large financial commitments to land-based disposal (e.g., on-site incineration, secondary waste treatment), made in response to EPA regulations. Ocean dumping would only be an option for drug companies that do not yet have secondary treatment.

2.6 REGULATORY FACTORS

Whether or not ocean dumping will become a more attractive disposal option in the future depends largely on its legal availability. Recent and proposed changes in federal, state, and local regulatory controls seem to favor consideration of the ocean for disposal of certain waste types. The most significant controlling regulations are examined below for their potential effect on ocean disposal.

2.6.1 The Marine Protection, Research, and Sanctuaries Act

This law and its regulations currently prohibit or strictly limit the ocean dumping of all types of material that would adversely affect human health, welfare, amenities, the marine environment, or economic potentialities. Amendments to the Ocean Dumping Act under consideration by the EPA would regulate the dumping of all materials into ocean waters that will result in degradation of the marine environment. It would encourage the removal of particular contaminants from materials before dumping occurs and prohibit the dumping of materials into any ocean waters until their effects have been adequately studied. This would not categorically exclude sewage sludge or industrial waste.

2.6.2 The Federal Water Pollution Control Act (Clean Water Act) and National Pollutant Discharge Elimination System (NPDES) (Sec. 402)

The National Pollutant Discharge Elimination System (NPDES) controls the discharge of pollutants from point sources. Discharges must comply with effluent guidelines, not violate applicable water quality standards and, in the case of ocean waters, must comply with Ocean Discharge Criteria (Sec. 403). Implementation of this program has resulted in the generation of treatment sludges that could be ocean disposed but are currently being disposed of in landfills, incinerated, or, in some cases, recycled and reclaimed.

2.6.3 Pretreatment (Sec. 307)

According to U.S. EPA regulations (40 CFR 403.5), municipalities operating Publicly Owned Treatment Works (POTW) must have an approved pretreatment program in place by July 1, 1983. A municipality must require all industrial dischargers using its facility to control particular pollutants in their discharge to the facility. Pollutants of concern are those which pass through or interfere with the treatment process. National pretreatment standards currently in effect prohibit the discharge of pollutants that: 1) create a fire or explosion hazard; 2) cause corrosive structural damage to the POTW, and in no case have a pH less than 5 (unless the POTW is specifically designed to accommodate low pH waste); 3) are solid or viscous enough to obstruct sewer flow or otherwise interfere with the POTW; 4) create a large oxygen demand and interfere with POTW operation; or 5) interfere with the biological treatment function of a POTW by addition of heat, and in no case create a temperature greater than 40°C (104°F) unless the POTW is designed to accommodate it.

Specific federal pretreatment standards for all priority industries are due to become effective by mid-1984. These standards are principally designed to require removal of pollutants that would pass through a POTW with essentially no treatment. Since municipalities are required to have their programs in place 6 months earlier, and these programs may be more stringent, whether or not the federal regulations take effect on the projected date may not be important.

The principal industrial sludges of interest that will be produced from pretreatment will be metal hydroxide sludges. The metals found in these sludges are generally prohibited from ocean dumping by the London Dumping Convention. No acids currently discharged to POTWs would comply with general pretreatment requirements. Increased amounts of acids and acid-neutralized sludges could be generated for potential ocean dumping as surveillance and enforcement programs succeed in eliminating these discharges.

Metal hydroxide sludges generated now by industries treating their waste to meet NPDES limitations, and generated in the future by pretreatment, could be retained for their metal value. Alternatively, these sludges, considered hazardous waste because of their metal content, could be solidified and dumped at sea. There have been recent proposals to stabilize hazardous waste by solidification and then landfill the solid material. Companies proposing such facilities have had a very difficult time siting them because, among other things, local communities do not want their towns to be the repositories for this material, stabilized or not. These materials may be suitable for disposal at sea, a burial ground that is perhaps more politically acceptable.

2.6.4 Waivers from Secondary Treatment (Sec. 301[h])

Section 301(h) provides for a waiver from secondary treatment requirements for a POTW provided that, among other things, such a waiver will not violate water quality standards, will not require reductions from other point sources to attain water quality standards, and that there is a pretreatment program in place and sources of toxic pollutants are controlled. Secondary sludge is not generated if only primary treatment is required.

Recommended amendments to the Clean Water Act by GAO (1981) would eliminate the time limit for making 301(h) applications. GAO recommends that the U.S. EPA revise its definition of Best Practicable wastewater treatment Technology (BPT) to allow primary discharges. (This is being considered for all municipalities [Inside EPA, 15 January 1982].) GAO also recommends that Step I facilities planning grant applications in coastal areas be required to consider discharging primary wastes into marine waters as an alternative to secondary treatment. GAO would revise the waiver application approach to consider separately communities by size and type of population served, making it easier for smaller communities with just domestic waste to obtain waivers. The EPA would provide small coastal communities with technical assistance in obtaining waivers. A recent court decision in Orange County, California, requires the EPA to consider applications to discharge sewage sludge and to make a determination (finding of fact) of whether there will be a significant adverse impact.

2.6.5 Ocean Discharge Criteria (Sec. 403)

Ocean Discharge Criteria are promulgated by 40 CFR 125.121 and require that all discharges of pollutants from a point source into the territorial seas, the contiguous zone, and the oceans must not unreasonably degrade the marine environment. "Unreasonable degradation of the marine environment" means 1) significant adverse changes in ecosystem diversity, productivity, and stability of the biological community within and surrounding the area of discharge; 2) threat to human health through direct exposure to pollutants or through consumption of exposed aquatic organisms; or 3) loss of aesthetic recreational, scientific, or economic values that is unreasonable in relation to the benefit derived from the discharge. Ocean Discharge Criteria do not materially affect, by themselves, the type or amount of ocean dumping that would be controlled by the MPRSA.

2.6.6 Resource Conservation and Recovery Act (RCRA)

The Resource Conservation and Recovery Act (RCRA) controls at the federal level the generation, transport, treatment, storage, and disposal of material classified by regulation as hazardous. Material that was formerly disposed of as solid waste or discharged to waste treatment plants without regard to potential toxic effects is now controlled and can no longer be discarded improperly. The effect of this law has been to identify increasing amounts of wastes that are hazardous and, because of the lack of suitable disposal sites, has resulted in pressure to have states try to identify proper hazardous waste disposal sites.

Siting acceptable hazardous waste disposal facilities has not been very successful. In Massachusetts, communities have successfully opposed siting such facilities within their borders. While publicly recognizing the need for proper facilities, communities have not been willing to accept them. Offshore disposal of solidified, detoxified hazardous waste may comply with ocean dumping regulations because the waste will no longer be toxic. Also, offshore disposal may be politically attractive because the burden of the disposal site is shared by all.

2.6.7 Safe Drinking Water Act

The Underground Injection Control Program of the Safe Drinking Water Act controls the injection of wastes into wells, to protect water supplies from contamination. Recent relaxation of EPA rules (July 1981) allows states implementing this program to exempt from protection certain underground water supplies, and to exempt oil shale and geothermal wells completely. Regulation of waste injection into wells might force waste disposers currently discharging to wells to seek other disposal alternatives, including ocean dumping. However, by designating certain subsurface areas for disposal and reducing the cost of such disposal, the need to seek alternatives is lessened.

2.6.8 Low-Level Radioactive Waste Policy Act

The Low-Level Radioactive Waste Policy Act (PL96-573) requires each state to be responsible for LLW wastes generated within its borders (except defense-related waste). States may enter into regional compacts, and, after January 1, 1986, compact states may refuse wastes from non-compact states. This act has clear implications for the number of shallow land burial sites. States where LLW waste generation is highest (e.g., the Northeast states) will have to develop disposal options long before 1986 in order to have them implemented by then. Massachusetts has responded by passing legislation providing for an investigation and study by a special commission relative to low-level radioactive waste. This commission is required to assess current laws and regulations, determine whether Massachusetts should participate in a regional LLW facility, and recommend a comprehensive process for siting a LLW facility.

2.6.9 State and Local Regulatory Factors

As state and local governments become increasingly aware of the problems associated with land disposal of solid and hazardous waste, more laws and ordinances are being enacted to control this activity. In many cases, this tightening of regulatory control at the state and local level is severely limiting disposal options for waste material, especially solid and hazardous waste. This tightening of control for land disposal, in combination with court rulings and proposed amendments to existing federal regulations that would force EPA to consider ocean disposal as a feasible alternative, tend to favor an increase in ocean disposal proposals for all types of solid waste. Some examples of these factors nationwide include:

- A ban on land disposal of sludge by the state of New Jersey.
- City of New York versus the U.S. EPA. Sewage sludge can be considered for ocean dumping by the EPA based on its actual effect on the marine environment.
- States being delegated federally mandated programs: RCRA, NPDES, Pretreatment, UIC – likely to be more strict.

- Orange County Court of Appeals decision. The EPA must develop a satisfactory procedure enabling municipal agencies to apply for 301(h) modified NPDES permits to put sludge in the ocean.
- Zoning and Home Rule may keep hazardous waste disposal facilities from being located on land.

2.7 COSTS OF ALTERNATIVE WASTE DISPOSAL METHODS

Waste disposal costs have been estimated and discussed by several investigators, and the results show considerable variation. Factors which contribute to inconsistent cost estimates include: the type(s) of waste considered; the geographic setting; the time frame of the analysis; the interest rate selected; whether indirect (social) costs are included; the volume of waste handled and plant capacity; and whether transportation/handling costs are included. A majority of the studies available have focused their attention on particular types of waste and the costs of alternative disposal methods for that waste. We will follow the same basic approach.

2.7.1 Sewage Sludge

The NRC (1977) study of sewage sludge management options provides a thorough discussion of factors which influence sewage sludge disposal costs and which can lead to disparate results for otherwise similar analyses. They also explicitly consider indirect costs and benefits but do not attempt to quantify these items. Table 2-9, taken from the NRC report, lists some of the types of direct, indirect, and social costs and benefits associated with sewage sludge management. Table 2-10 presents cost estimates for alternative sewage sludge disposal options as presented by the NRC.

Each disposal option exhibits significant variation in direct costs, traceable in part to differences in land, labor, energy, and other costs at particular facilities. The ocean disposal option appears to present a relatively attractive method in terms of direct costs. Other options, including incineration and land application, offer certain indirect benefits as well (heat recovery and soil enrichment, for example) which could improve their relative standing with regard to costs.

2.7.2 Dredged Material

The critical costs associated with dredged material disposal are transportation costs. Indirect and social costs, potentially important in areas of intensive fishing or recreational activity, and indirect benefits associated with beach replenishment, have not been quantified in the available studies. The U.S. Army Corps of Engineers (1979) presents cost data for dredging projects in the New York District for the 1972-1976 period. The costs presented include both dredging and disposal and range from \$1.32 to \$1.44 per cubic yard of material dredged. Transport costs represented roughly 70% of this unit cost. Similar estimates from other sources range as high as \$2.28 per cubic yard for ocean disposal and \$3.05 for land disposal.

TABLE 2-9. COST AND BENEFIT ITEMS FOR CONSIDERATION.

Types of direct, indirect, and social costs and benefits associated with sludge management are illustrated, but this list is not exhaustive. In addition to the water quality benefits attained by wastewater treatment, there are several potential benefits which may accrue from the use of sludge. A review of information presented here could aid in arriving at a decision concerning sludge management. (From NRC, 1977.)

Cost and Benefit Items for Consideration	Direct	Indirect	Social
Cost			
Capital Expense: treatment, disposal, reclamation	X		
Operating Expense: treatment, disposal, reclamation	X		
Land Values: proximate to treatment site		X	
Health Hazards: proximate to treatment site		X	X
Aesthetic Values: proximate to treatment site			X
Transport Costs: treatment site to disposal or reuse site	X		
Acquisition Costs: land for disposal or reuse	X		
air for disposal			X
water, e.g., ocean, for disposal			X
Land Values: proximate to disposal site		X	
Health Hazards: proximate to disposal site		X	X
Aesthetic Values: proximate to disposal site			X
Productivity Cost: labor and capital for agriculture		X	
Governmental Cost: regulatory, research, etc.		X	
Raising and Disbursing Tax Monies		X	
Effects on Fisheries		X	
Industrial Wastewater Pretreatment	X	X	
Benefit			
Revenues from fertilizer sales	X		
Revenues from other reclaimed products	X		
Proceeds from sales of land reclaimed	X		
Land values adjacent to reclaimed site		X	
Value of goods produced on reclaimed land		X	
Use of treated water produced in course of generation of sludge		X	X
Ocean productivity		X	
Alternate employment for fertilizer, labor, and capital		X	

TABLE 2-10. COST ESTIMATES FOR DISPOSAL OF SEWAGE SLUDGE BY VARIOUS OPTIONS. (1976$/DRY TON) (From NRC, 1977.)

Pyrolysis	Wet Oxidation	Incineration	Drying	Oil Dehydration	Composting	Landfill	Land Application	Lagoon	Ocean Disposal
48-88	59-113	33-173	60-134	41-116	22-177	27-141	46-160	138	9-61

2.7.3 Coal Ash and Flue Gas Desulfurization (FGD) Sludge

Estimates of FGD sludge disposal costs reported in the literature and summarized by EPRI (1979) range from $1 per dry ton (ponding after fixation) to $25 per dry ton (dry disposal with fixation) in 1977 dollars. Unfortunately estimates in different studies are difficult to compare because of inconsistencies in assumptions. EPRI reports operating data for four power plants that indicate FGD sludge disposal costs of $8.53 to $22.47 per dry ton. Table 2-11 summarizes cost estimates for several FGD sludge disposal methods.

TABLE 2-11. COST ESTIMATES FOR DISPOSAL OF FGD SLUDGE BY VARIOUS OPTIONS. (1977$/DRY TON.) (From EPRI, 1979.)

Wet Disposal in Ponds with Artificial Liner	Wet Disposal in Ponds with Natural Liner	Wet Disposal in Ponds with Fixation	Dry Disposal without Fixation	Dry Disposal with Fixation	General Disposal with Fixation
3.50-18.00	4.00-5.62	1.00-17.90	3.70-11.00	8.30-25.00	5.00-15.00

Disposal costs for coal ash have been estimated by EPA (1979) in an attempt to determine the impacts of RCRA on the waste handling procedures of electric utilities. Their methodology involved a consideration of current disposal practices and costs, in conjunction with incremental cost increases brought about by RCRA-mandated design changes. The analysis also assumes that ash and FGD sludge are combined prior to disposal.

The disposal cost for the ash/sludge mixture was estimated to be $7.51 per metric ton for an unlined pond, and $3.00 per metric ton for landfilling. Incremental costs resulting from RCRA-required treatment or containment measures were estimated to be $8.23 to $9.47 per metric ton for liners and $6.32 per metric ton for chemical stabilization. The total ash/sludge mixture disposal costs were estimated to fall in the $3.00 to $17.00 per metric ton range depending on the degree of containment and pre-treatment implemented.

2.7.4 Waste Gypsum

Most gypsum by-product is stockpiled at the sites where the material is produced, often in wet ponds. The distribution of waste gypsum among disposal options and commercial uses is not known, nor are the costs associated with the various end uses. However, the costs of wet ponding gypsum sludge are probably similar to those of wet ponding FGD sludge. These cost estimates are discussed in subsection 2.7.3.

2.7.5 Waste Acids

In 1974 GAF Corporation examined alternative methods for disposing of a waste product consisting of sodium chloride, sodium sulfate, and several species of nitrobenzoic acid (Klein, 1974). The methods examined included deep-well injection, incineration/deep-well injection, and ocean dumping, with estimated disposal costs of $1.71, $6.46, and $8.33 per ton, respectively. These estimates have probably doubled since the time the GAF report was prepared.

Neutralization of waste acid at a titanium dioxide pigment manufacturing plant has resulted in a $95 per metric ton increase in the cost of producing the pigment (Trees et al., 1979). This cost was not presented on a dollar per unit of waste basis, nor was the disposal/storage cost of the by-product gypsum estimated. If it is assumed that each ton of product yields 2.4 tons of waste acid, a treatment cost of $39.58 per ton of waste acid can be derived. Most of the resulting gypsum has been stockpiled on-site, and a small portion has been sold for wallboard manufacture. Estimates of the associated storage costs and sales revenue were not presented in the report.

Acid wastes are typically ocean dumped in bulk form, which is less costly than disposal of other industrial wastes in containerized form. Reed (1975) presents cost estimates for bulk ocean dumping of industrial wastes which do not include treatment, storage, loading, monitoring, and any associated indirect or social costs. For the U.S., the disposal cost averaged $1.70 per ton and ranged from $0.60 to $9.50 per ton in 1968, and is probably much closer to $30 to $60 per ton today.

2.7.6 Low-Level Radioactive Waste (LLW)

Transportation costs are an important consideration in LLW disposal because of the limited number of disposal sites available in the U.S. The DOE (1980) reports disposal costs at two existing LLW disposal facilities to be in the $6.00 to $7.75 per cubic foot range, exclusive of transportation costs. Shipping costs for a standard (1140 cubic foot) truck load of material would add approximately $1.50 per mile, while shipping costs for material shipped in special protective containers (300 cubic foot) would add roughly $3.00 per mile to the disposal fee. Material shipped 1000 miles by these methods would result in total disposal costs of approximately $7.32 to $10.75 per cubic foot.

The same report estimates disposal costs for 1985 assuming centralized disposal sites, thereby reducing the transportation cost share. The analysis includes costs of land, buildings, equipment, licensing, operation and maintenance, disposal fees, and transportation. The per cubic foot disposal cost estimate ranges from $13.87 to $66.16 depending on the site location and handling capacity. Estimates are also provided for incineration of low activity materials prior to landfilling at appropriate sites. These cost estimates range from $7.65 to $16.01 per cubic foot, depending once again on site location and capacity.

2.7.7 Seafood Processing Wastes

Unfortunately, cost data are unavailable for disposal of seafood processing wastes. In many cases, the wastes are discharged after some level of treatment, or ocean dumped. The ocean dumping methods and costs are probably well represented by those of sewage sludge.

2.7.8 Industrial Hazardous Wastes

Hansen and Rishel (1981) and the present program obtained cost estimates for disposal of a variety of "industrial hazardous wastes." These disposal costs exhibit significant variation depending on the composition of the material being disposed of and the disposal technique being considered. Table 2-12 summarizes this cost information.

TABLE 2-12. COST ESTIMATES FOR DISPOSAL OF INDUSTRIAL HAZARDOUS WASTES BY VARIOUS OPTIONS. ($/TON)

	Hansen and Rishel (1978 $)	Present Program (1982 $)
Incineration	537-566	100-3000
Land Disposal	111-341	20-500
Chemical Fixation with Solids	199	50-100
Chemical Fixation without Solids	53	50-100
Encapsulation	95-103	----
Deep-well Injection	----	40-70
Chemical Treatment/Sludge Disposal	----	25-200

The estimates for incineration cover the largest range, with PCB incineration being most costly. Land disposal costs also show a wide range of estimates, with chemical landfilling of drums being the most costly. As with other waste categories, however, these cost estimates fail to consider external (social) costs which can be of critical importance in waste handling decisions.

3. ENVIRONMENTAL AND PUBLIC HEALTH IMPLICATIONS OF LAND-BASED WASTE DISPOSAL ALTERNATIVES

This section of the report summarizes information on the environmental and public health implications of land-based alternatives. The information indicates that no alternative is risk-free, although some alternatives pose greater risks than others. We have not attempted to quantify the relative risks of various alternatives as this is beyond the scope of our present project. However, we believe the information presented does underscore the need for a cross-media risk analysis approach to waste management. No alternative should be rejected outright without consideration of the risks to health and the environment posed by other alternatives.

Several cross-media risk analyses are presently underway. The EPA's Sludge Policy Committee has contracted with Booz Allen & Hamilton Inc. to gather information that can be used in a cross-media analysis of sewage sludge disposal. The EPA's office of Solid Waste Management has contracted with ICF Inc. to examine the public health and environmental implications of various methods of hazardous waste disposal. In addition, New York City (New York) and the South Essex Sewage District (Massachusetts) have conducted cross-media risk analyses on their sewage sludges. Undoubtedly, other examples of the application of this approach could be found.

We present below discussions of environmental and public health implications of several land-based disposal alternatives: landfills, land burial, land spreading, incineration, solidification, and deep well injection. This review is not meant to be exhaustive, but rather to provide background information on the kinds of risks associated with these alternatives.

3.1 LANDFILLS AND LAND BURIAL

Sanitary or hazardous waste landfills have been used for disposal of a number of waste types including sewage sludge, dredged material, fly ash, FGD sludge, industrial sludge, and low-level radioactive wastes (shallow land burial).

3.1.1 Sanitary Landfills

Depending upon its design and location, a wide range of wastes can be deposited in sanitary landfill. The amount of waste it is capable of receiving is governed by site size and available equipment. Leachate is controlled by using an impervious liner, made of either clay or plastic, and a drainage and collection system that returns the leachate for treatment.

The landfill option is fairly straightforward with relatively low initial and operating costs. Land use is an important consideration, for once the land is used for this option, it cannot be readily used for other purposes. As land is consumed for landfill, the availability of land in the community will decrease, and while this will not become a pressing problem only because of landfill, it will become more acute. It has also been suggested that landfills may require hundreds of years of maintenance due to settlement, methane-gas generation, and leachate formation (Stone, 1977).

A major concern when employing this option for waste disposal is that many landfills are not operated properly, leaving unstable cells to collapse, causing slides and exposing refuse. This usually is not a dramatic slide, but rather a small opening through which insects and rodents can enter with the possibility of picking up and spreading diseases. Pollution of surface and groundwater due to runoff and leachate may also have long-term effects on the environment and human health.

Leachate contains high concentrations of organic and inorganic compounds and is a result of percolation of water through landfill. It is highly variable in nature depending on the characteristics of waste material; the chemical and physical conditions of landfill, such as temperature, pH, redox potential, moisture, and age; the quality and quantity of external sources of water; and, the composition of covering soils.

After the waste is placed in a sanitary landfill, it undergoes a number of biological, physical, and chemical changes simultaneously. The biological activities depend, to a large extent, on the composition of the waste and the moisture content in the landfill (Eliasson, 1972). The concentration of free oxygen, on the other hand, determines the mode of decomposition. It has been reported that in aerated landfills, the rate of refuse settlement was approximately five times greater than that for the anaerobic landfills. The former have higher internal temperatures and greater rates of waste stabilization (Merz and Stone, 1969). Because landfill cells are generally sealed by compacting, anaerobic decomposition of organic matter is the prevailing reaction. As a result, large quantities of organic acids are formed. Methane, carbon dioxide, hydrogen sulfide, and hydrogen are the major constituents of the gaseous products. Solubilization of organic compounds by ionization, formation of hydrogen bonding, and hydrolysis contributes to the leachate enrichment in dissolved solids.

Due to the anaerobic condition, the redox potential generally favors reduction. Thus, elements are more stable in the reduced state. Reduction of insoluble Fe^{3+} and Mn^{4+} to the soluble forms of Fe^{2+} and Mn^{2+}, followed by the reduction of nitrate, ferric hydroxide, alcohol, carbohydrate, sulfate, sulfur, and carbon dioxide, not only lead to an increase of hardness and alkalinity in leachates, it is also a source for groundwater contamination. Furthermore, weathering reactions such as dissolution of calcite, apatite, and alumino-silicates by carbonic acid, produced through the solution of CO_2 from landfills in percolating water, also contributes to the mineral contents of the leachate. Since minerals continue leaching for indefinite periods of time, they present more of a problem than do organic leachates (Chen and Bowerman, 1974).

Due to the great variance in the elements contributing to leachates formation, their compositions vary widely. The concentration range of components found in leachates and in well water adjacent to sanitary landfills in the Los

Angeles Basin was reported by Chen and Bowerman (1974). It was found that leachates with a high concentration of total hardness were generally low in trace metals (lead, zinc, and cadmium) but contain high concentrations of ammonia and have a high chemical oxygen demand (COD). The pollutant concentration ranges in leachate for typical domestic refuse have been reported by Stone (1977). It was suggested that landfill leachates may contain dissolved and suspended materials of all sorts within these limits (Table 3-1).

TABLE 3-1. CONCENTRATION RANGE OF POLLUTANTS IN LEACHATE FROM TYPICAL DOMESTIC REFUSE. (Stone, 1977)

Ion	Concentration Range (mg/l)
phosphate	5-130
sulfate	25-500
chloride	100-2400
sodium	100-3800
nitrogen	20-500
hardness (as $CaCO_3$)	200-5250
chemical oxygen demand	100-51000
total residue	1000-45000
iron	200-1700
zinc	1-135
nickel	0.01-0.80
copper	0.10-9.0
pH	4.0-8.5

The primary transport medium for leachate products is the groundwater flow system which is governed by hydrogeologic properties of the earth materials, hydrology of groundwater, and pumping of wells in the area. Due to the low flow velocities and low diffusion rates in groundwater reservoirs, serious consequences may result when leachate pollution is present (Chen and Bowerman, 1974). Pollution of surface water can also lead to contamination or eutrophication of the water body.

Deterioration of air quality can result from organic chemicals evolved from sludges after landfilling. The type and quantity of volatile chemicals emitted depend largely on the nature of the waste, pH, the oxidation-reduction potential, the moisture content, and the temperature of the soil environment. Certain of the compounds from sewage sludge have been identified (Rains et al., 1973). These include alcohols, carbonyl-containing compounds, nitrogen-containing compounds, sulfur-containing compounds, and simple organics.

The chemicals which contain sulfur, reduced nitrogen, and carbonyl moieties are malodorous. Additional malodorous compounds may be contributed by the microbial methylation and volatilization of inorganic elements such as selenium, mercury, arsenic, lead, and tellurium (Becker et al., 1974; Doran, 1976; Doran and Alexander, 1975; Wong et al., 1974; Wood, 1974). Methylation is an essential metabolic process of all biological systems and certain of these elements are indigenous to soil. However, the methylated forms of these elements are greatly increased by landfilling of wastes such as sewage sludge due to an increase in both carbon and metal availability.

Although there is no apparent correlation of toxicity and odor of gases, nor is there any definite relationship between odor of gases and specific organic diseases, malodors have been frequently identified as air pollutants of greatest concern by public surveys (Osag and Crane, 1974). In addition to their adverse economic effects on the growth and development, property values, and tax revenues of a community (Sullivan, 1969), physiological responses, some of them of subjective nature, may also be induced by malodors (Mosier et al., 1977). These include poor appetite, lower water consumption, impaired respiration, nausea and vomiting, insomnia, mental stress, and allergic responses.

Dust emission is also frequently associated with landfills. The nature and quantity of the background dusts are conceivably a function of the types of covering soils and their susceptibility to wind erosion. Depending on the composition of disposed waste, various organics, volatile compounds, and microorganisms may be present on these particles.

3.1.2 Hazardous Waste Landfills

Much of the previous discussion concerning general risks associated with sanitary landfills applies to hazardous waste landfills. However, a few key points are emphasized here.

Most of the operating hazardous waste landfills in the United States isolate their contents by combinations of the following: geologic isolations, natural or synthetic liners, and leachate collection. The protection of groundwater from leachate generated by mobile liquids in the landfill or the interaction of waste with rainfall, groundwater, or surface water is of prime concern.

The operating chemical landfills are almost exclusively designed with clay soils used as liners or barriers. The suitability of natural and compacted clay soils used for lining of waste management facilities has recently come under close scrutiny by the EPA. Two recently published papers have suggested some problems with increased permeation of organic fluids through clay soil liners (Green et al., 1981; Anderson et al., 1981). The permeabilities of clay soils in contact with solvents such as carbon tetrachloride and xylene showed rapid increases that may be due to structural changes in the soil. Solvent breakthrough phenomenon was observed in the number of solvent/clay-soil systems. Benzene, xylene, and carbon tetrachloride all were observed to cause shrinking and cracking of clays resulting in bulk permeability or channel flow through the clay liner.

The implication of these studies is that the state-of-the-art hazardous waste landfills will leak organic leachate faster than predicted. An unreleased Princeton University study of leakage at four secure chemical landfills was referenced during testimony given to the House Government Operations Subcommittee on November 4, 1981. The study showed that "significant leakage" from these facilities had occurred.

Additional documentation of the potential ineffectiveness of clay liners was a December 22, 1981 St. Louis Post-Dispatch report of the unexpected migration of leachate at the SCA Landfill in Wilsonville, Illinois. State environmental officials announced the finding of organic solvents in a disposal site well. Permeability calculations predicted the solvents would have taken 500 years to reach this well. The material, however, had been buried in mid-1978 and had migrated to the well in only 42 months.

Concern over the control of leachate from hazardous wastes disposed of in landfills has generated some new controls on this disposal activity. California has enacted legislation that will ban the land disposal of six types of "high priority wastes" beginning January 1, 1983. Those wastes are: PCBs, pesticides, toxic metals, cyanides, halogenated organics, and non-halogenated organics. The California Office of Appropriate Technology's report, "Alternatives to the Land Disposal of Hazardous Wastes: An Assessment for California," estimates that these six categories of wastes account for 40% of all wastes landfilled in the state. The report proposed that "it is technically feasible to recycle, treat, or destroy approximately 75% of all the hazardous wastes now disposed of in Class I and II-1 landfills." The concern with these classes of wastes is again the eventual migration and persistence of leachate from landfills.

Estimates of between three and six new treatment and incineration facilities will be required in order to safely manage California's high-priority wastes. It should be noted though that recovery, incineration, or detoxification facilities will still require ultimate land disposal for residues from these operations. Additionally, there are groups of wastes, such as spent fluorinated solvents generated by the semiconductor industry, that cannot be recycled or incinerated because of severe corrosion caused by the formation of hydrofluoric acid during thermal decomposition.

3.1.3 Shallow Land Burial of Low-Level Radioactive Waste

Current disposal practices involve waste volume reduction, packaging, and ultimately shallow land burial. Liquids from power plants are treated by evaporation, filtration, and ion exchange. After evaporation of water, the concentrated liquid is solidified with a binding agent like cement. Filtration is used to remove contaminated suspended solids from water and ion exchange is used to remove dissolved radionuclides from solution. Institutional wastes (generally from academic and medical centers) consist mainly of organic liquids and are mixed and packed with vermiculite, a process which increases volume.

Solids are generally compacted. After volume reduction, wastes are packaged in metal drums or tanks, fiberglass reinforced plywood boxes, or, in the case of dry trash, strong, tight cardboard or wooden boxes.

Present disposal practices and the status of commercial and government shallow land burial was summarized by the National Low-Level Waste Management Program (1980) from which the following is taken.

Commercial low-level wastes currently can be received at three commercially operated disposal sites. These shallow land burial facilities were all open before 1971; the criteria applied to the establishment of these sites reflected the knowledge of that time. Current criteria for site selection include environmental factors such as soil types, groundwater conditions, terrain, annual precipitation, and geologic features. Other criteria refer to population concentrations, transportation access, current and future land use, and general location.

Wastes are buried in shallow trenches, typically 40 feet wide at the top, 25 feet wide at the bottom, 20 feet deep, and 600 feet long. Waste containers are placed into the trenches and covered daily with about 4 feet of dirt. Another 2 to 6 feet of soil, some of which may be compacted clay, is placed on top to form a cap. After the trenches are filled and capped, their locations are marked with permanent stone or metal markers indicating the locations as well as the volume and radioactivity of the buried material. Records are kept to verify the placement and location of waste shipments.

Trenches are constructed as close together as possible, usually about 20 feet apart, depending on the slope. The local terrain may force less efficient use of land if hills or ravines are present. Space is also reserved around the perimeter of the trench area for a buffer zone.

Shallow land burial has been used for the disposal of low-level wastes since the 1940s. At first, burial grounds were operated by the U.S. Atomic Energy Commission, a predecessor of the Department of Energy. The first commercial disposal site was opened near Beatty, Nevada, in 1962. By 1971, six commercial sites had been licensed to dispose of low-level radioactive wastes. Since 1975, three of these six sites have closed. The West Valley and Maxey Flats sites closed in 1975 and 1977, respectively, as a result of operational problems related to water management. Because of poor trench design and site selection, rainwater collected in the trenches and became contaminated with radionuclides. The rainwater had to be collected and processed to protect groundwater and surface-water systems.

Water management problems persist at both sites despite their closure. At Maxey Flats, the trench water continues to be processed in an evaporator. Remedial programs are underway at West Valley to minimize water infiltration into the trenches and provide for surface runoff. The only radionuclide that has migrated off the site at both locations is tritium. Tritium, which becomes part of the water molecules, has traveled limited distances beyond the site borders through surface and atmospheric waters. It has not been detected off the site in groundwater systems. At Maxey Flats, unexpected migration of other radionuclides has also been restricted to the site.

In 1978, the available trenches at the Sheffield site were filled, and the site operator applied to the Nuclear Regulatory Commission for a license for additional disposal space. Action on the application was delayed; hence, the site was closed in March 1979 when the site operator withdrew the application. While Sheffield had not experienced water management problems like the other two closed sites, geologic evidence indicated a possible hazard. Tritium has now migrated in groundwater a few meters off the site at Sheffield.

The three sites that remain open are the Barnwell, Beatty, and Hanford facilities. Possible restrictions on the volumes of wastes accepted at Barnwell and Hanford could divert large volumes of wastes to the Beatty site, causing that site to be filled in the mid-1980s. The disposal capacity at the remaining two sites would then be insufficient to handle all low-level radioactive wastes. The disposal capacity at Department of Energy sites appears sufficient for the near future. It has been suggested that some of this space be used for commercial waste.

MacBeth et al. (1979) considered the environmental impacts associated with shallow land burial. There are both radiological and non-radiological impacts. Non-radiological impacts include routine hazards to construction workers as are found in the construction and surface mining industries. Radiological hazards include exposure during transportation and handling of wastes by site workers. Currently waste disposal workers at burial sites do not receive doses in excess of Department of Transportation guidelines.

Long-term hazards include exposure of future reclamation workers, contamination of agricultural land during future site excavation, and groundwater contamination. In considering hazards due to future reclamation, the future is considered as 150 years after disposal, since it would take strontium-90 (one of the longest lived of the common radionuclides in LLW), 150-300 years to decay to a level of acceptable hazard (National LLW Management Program). MacBeth et al. (1979) consider that if a site is reclaimed in 150 years, the dose to workers due to inhalation of dust will be 110 millirems (mrems)[1] over 500 hours. A dose of 625 mrems/year would be encountered by an individual who consumes 10% of his food from a dust-contaminated field. If an individual were to obtain 100% of his drinking water from a well on-site 12 years after disposal, he would be exposed to 80 mrems/year. After 150 years, decay, dilution, and adsorption would reduce this figure to a level similar to that for nearby off-site wells.

The Nuclear Regulatory Commission (NRC) (MacBeth et al., 1979) has evaluated alternative methods for the disposal of low-level radioactive wastes. The alternatives evaluated include: shallow land burial (presently practiced), improvements to shallow land burial, deeper burial, mined cavity disposal, disposal in structures, and ocean disposal. At present, only shallow land burial is being used in the U.S. Each alternative was evaluated on the basis of nine factors. These included factors associated with technological status (compatibility with waste, site selection, safeguards, environmental effects, availability of techniques); factors associated with sociopolitical acceptability (institutional control, public acceptance); and factors associated with economic feasibility (individual consumer costs, industrial costs). Improvements to shallow land burial involve better disposal trench capping, improved trench design, better operational and water management techniques, improved waste forms, and in situ encapsulation of buried wastes. Deeper burial is similar to shallow burial except that the tops of the disposal trenches would be 10-15 meters below the surface. Disposal in mined cavities (natural features or mined caverns) would reduce hazards associated with reclamation and contamination of groundwater. Disposal in structural facilities of reinforced concrete would provide an added barrier to release. The structure may be an exposed structure or may be earth covered. Table 3-2 presents a weighted comparison of alternatives

---

1. Roentgen equivalent man (mammal) (rem) This unit is the quantity of ionizing radiation such that the energy imparted to a biological system per gram of living matter by the ionizing particles present in the locus of interest has the same biological effectiveness as one rad of 200 to 250 kilovolt X rays. The unit was proposed and named by H.M. Parker about 1950. A millirem is 1/1000 of a rem.

TABLE 3-2. WEIGHTED COMPARATIVE ANALYSIS FOR ALTERNATIVES. (After MacBeth et al., 1979)

	Evaluation Factors									
	Compatibility with Waste	**Site Selection**	**Safeguards**	**Environmental Effects**	**Availability of Techniques**	**Institutional Control**	**Public Acceptance**	**Consumer Costs**	**Industrial Costs**	**Weighted Comparison***
Weight	0.08	0.12	0.06	0.11	0.10	0.11	0.16	0.14	0.12	
Alternatives										
Shallow-Land Burial-Eastern Site	0.08	0.12	0.06	0.11	0.10	0.11	0.16	0.14	0.12	1.0
Shallow-Land Burial-Western Site	0.08	0.11	0.06	0.19	0.10	0.11	0.16	0.29	0.25	1.4
Improved Burial-Eastern Site	0.08	0.11	0.06	0.10	0.10	0.11	0.14	0.14	0.12	0.96
Improved Burial-Western Site	0.08	0.11	0.06	0.18	0.10	0.11	0.14	0.29	0.25	1.3
Deeper Burial-Eastern Site	0.08	0.14	0.05	0.07	0.11	0.12	0.13	0.15	0.13	0.98
Deeper Burial-Western Site	0.08	0.13	0.05	0.14	0.11	0.12	0.13	0.31	0.26	1.3
Abandoned Mine-Eastern Site	0.08	0.18	0.05	0.08	0.12	0.13	0.13	0.15	0.13	1.1
Abandoned Mine-Western Site	0.08	0.17	0.05	0.15	0.12	0.13	0.13	0.32	0.28	1.4
New Horizontal Shaft Mine-Eastern Site	0.08	0.17	0.05	0.09	0.13	0.13	0.11	0.21	0.18	1.2
New Horizontal Shaft Mine-Western Site	0.08	0.16	0.05	0.17	0.13	0.13	0.11	0.38	0.32	1.5
New Vertical Shaft Mine-Eastern Site	0.08	0.16	0.05	0.09	0.13	0.13	0.11	0.22	0.19	1.2
New Vertical Shaft Mine-Western Site	0.08	0.14	0.05	0.17	0.13	0.13	0.11	0.38	0.32	1.5
Above Grade Structure-Eastern Site	0.08	0.11	0.07	0.14	0.11	0.12	0.14	0.53	0.46	1.8
Above Grade Structure-Western Site	0.08	0.10	0.07	0.22	0.11	0.12	0.14	0.67	0.58	2.1
Buried Structure-Eastern Site	0.08	0.11	0.07	0.15	0.11	0.12	0.14	0.56	0.48	1.8
Buried Structure-Western Site	0.08	0.10	0.07	0.22	0.11	0.12	0.14	0.71	0.61	2.2
Direct Ocean Dumping	0.06	0.14	0.03	0.20	0.10	0.13	0.22	0.42	0.36	1.7
Ocean Projectile Disposal	0.08	0.17	0.03	0.20	0.13	0.13	0.19	1.27	1.09	3.3

* Weighted Comparison is the sum of the weighted evaluation factors for each alternative. Higher values indicated less desirability.

(including ocean disposal). Shallow land burial and improved or deeper land burial are the most favorable options on the basis of the NRC analysis (MacBeth et al., 1979).

3.2 LAND SPREADING

Land spreading is used for disposal of sewage sludge and for certain kinds of industrial wastes. Land disposal of industrial sludge and wastewater by mixing into surface soil is called soil incorporation or landfarming.

3.2.1 Land Spreading of Sewage Sludge

The application of sewage sludge on agricultural and forest land primarily serves as a means of disposal. However, the sludge can benefit the land by improving soil physical properties and providing it with essential plant nutrients. In addition to its high organic contents, sludge contains the three major plant nutrients, nitrogen (N), phosphorus (P), and potassium (K), but at lower levels than in commercial fertilizers. If managed properly, sludge can meet a portion of the N, P, and K requirements of crops.

Different methods can be used to handle sewage sludge depending on its solids content. Sludge with 25 to 30% total solids are handled as a solid (e.g., by shovel or fork). Sludge of up to 10% solids content can be pumped with special equipment, while sludge slurries with total solids up to 5 to 6% may be applied with field sprinklers (McCalla et al., 1977). By adjusting sludge solids content, most farmers may possess the required equipment for sludge handling and application. Therefore, the farmers will absorb the operational and maintenance costs of land spreading machinery. With this in mind, the economics of land spreading is very interesting compared to other high-cost alternatives. The major cost would be in a suitable pretreatment process for the raw sludge. Although the raw sludge from the treatment plant can be spread directly on land, such a practice is undesirable due to the possible existence of pathogenic organisms and odor problems.

Land spreading is not a new concept. Originally it was a major alternative to dumping raw sewage into waterways. The raw sewage was poured over land and its water content remained in the soil. The difficulty in locating suitable land sites, the constant dwindling of freshwater resources, and the ecological problems and health hazards associated with such a disposal method have resulted in the abandonment of this practice. Currently, there is renewed interest in applying sewage sludge to farmland due to three factors: the increasing amount of sludge; the desire to utilize "wastes" as renewable resources; and application on farmland is generally the lowest cost option for ultimate sludge disposal.

To minimize the potential adverse environmental impacts and health hazards, pretreatment of sewage sludge prior to its land application is necessary. A number of methods can be used for this purpose. The most common ones are composting, aerobic digestion, anaerobic digestion, and lime or chemical treatment.

The comparative effectiveness of conventional municipal sludge stabilization processes for reducing density levels of indicator and pathogenic organisms has been evaluated by Pederson (1981). It was found that:

A. Mesophilic composting reduces total coliform and fecal coliform density levels by more than 3 and 4 logs, respectively. Salmonella densities are generally reduced by 1 to 3 logs, resulting in less than 10 organisms/gram dry weight of sludge. The numbers of Shigella sonnei, Staphylococcus aureus, and Serratia marcescens are also significantly reduced. Most viruses of concern are vulnerable to composting. However, fecal streptococcus and Mycobacterium tuberculosis are quite resistant and the pathogenic fungus Aspergillus fumigatus thrives under the mesophilic composting conditions.

B. Aerobic digestion appears to be ineffective in reducing the number of pathogenic organisms in sludge.

C. Anaerobic digestion consistently results in an approximately 10- to 100-fold reduction in density of indicator and pathogenic bacteria and viruses.

D. Lime treatment also results in a 10- to 1000-fold reduction of indicator and pathogenic bacteria.

E. Ova and cysts of parasitic tapeworms, flatworms, and roundworms are quite resistant to these sludge treatments.

Soil is the domain of myriads of microflora and micro- and macro-fauna. The intense competition among these organisms for the available nutrients are important mechanisms helping to control pathogens that may be introduced to soil as a result of sludge application. Their biochemical activities, together with the chemical and physical actions of the soil, also transform, solubilize, and/or immobilize the organics and inorganics of the sludge resulting in an increase or a decrease of their availability as plant nutrients and as environmental pollutants.

Application of sludge to soil, in addition to increasing its plant nutrient contents, also increases soil organic matter content, which improves both the soil chemical properties by increasing its cation exchange capacity (CEC) and the soil physical properties by increasing the percentage of its water-stable aggregates and its moisture retention capacity (Epstein, 1975; Epstein et al., 1976). CEC is one of the most important soil characteristics affecting the availability of nutrients and essential elements to plants and the leaching of toxic metals and certain other pollutants. It also provides the soil with a pH-buffering capacity. In fact, CEC, together with the measured pH (active acidity) and the degree of saturation of the exchange complex with Al^{3+} and $Al\ (OH)^{2+}$ (reserve acidity), are the factors governing the true soil acidity (Keeney and Wildung, 1977). Water, on the other hand, is a key constituent affecting practically every physical process within the soil (Letey, 1977) and is one of the most significant factors influencing the growth and the biochemical activities of soil micro-organisms. Increased water retention could also decrease water stress in crops during the growing season.

It has been estimated that the plow layer (about six inches deep) of an acre of soil weighs approximately 2×10^6 pounds. Thus, adding 20 tons of sludge with a 50% dry weight organic matter would increase the surface soil organic matter by 1% (Christensen, 1977). However, the actual increase depends upon the balance between the added organic matter and the rates of decomposition by soil microorganisms.

Major organisms of health concern that may be present in sewage and sewage sludge are presented in Tables 3-3 and 3-4. The survival times of certain pathogens in soil and on plant surfaces are shown in Table 3-5 (Doran et al., 1977). Similar results have been reported by a number of investigators. Beard (1938) and Mallmann and Litsky (1951) have found that S. typhosa could survive up to 85 days in soil, but with soils which have poor moisture-retaining power, survival in periods of drought may be as brief as two days. Mycobacteria, on the other hand, can survive dry conditions in soil for more than 150 days and survival times as long as 15 months have been reported (Greenberg and Kupkan, 1957). While protozoan cysts are sensitive to dessication, Ascaris and Ancylostoma ova remain viable for long periods and have withstood conditions where the moisture content of soil was less than 6% and temperatures above 40°C for 60 to 80 days (Cram, 1943). Viable Ascaris eggs have been recovered up to 170 days under more favorable conditions. Little definitive information on the survival of virus in soil exists, but one could expect it to be of the same order of magnitude as survival in wastewater where persistence as long as 100 days has been reported (Akin et al., 1971).

The movement of bacterial and viral pathogens in percolation and runoff has been extensively reviewed in the literature (Krone, 1968; Krone et al., 1958; Romero, 1970). Generalizations are difficult but it appears that viral and bacterial movement in soils is related directly to the hydraulic infiltration rate and inversely with media particle diameters. Factors influencing bacterial and viral inactivation in soil, such as oxygen tension, temperature, and the presence of competing organisms and antimicrobial agents, will also be determining variables. Other factors that have an influence on adsorption phenomena in soils such as pH value, multivalent cation concentration, and clay content of the soil, will influence removal of pathogens, particularly virus, in soils. The majority of the studies reviewed indicate that the upper layers of soil are most efficient in removing bacteria.

It should be noted that pathogens removed in the upper layers of soil will be concentrated near the soil area where crops will be grown. Pathogens could directly pass to crops such as lettuce or beets from soil contamination or be carried to above-ground crops by flies or dust. The dustborne epidemic of Salmonellosis in Israel indicates that this is not outside the realm of possibility. Fractures or channels in underlying geological formations will also affect pathogen movement in soil. Vogt (1961) reported a waterborne epidemic of infectious hepatitis that followed limestone fractures from a septic tank to individual wells considerable distances away from the source. Laboratory and field studies cannot be completely relied upon as to distance of travel since small quantities of pathogens may escape removal, particularly in fractured geological formations. Epidemiological evidence of bacteria and viruses traveling considerable distances to water supplies is discomforting, particularly since levels of pathogens sufficient to result in infection may not be detectable in dilute solution with presently available techniques. Since the survival time of the organisms may be quite long in soil, continuous application of sludge onto soil could result in an accumulation of pathogens. An equilibrium value could be reached where die-off and removal through percolation and runoff are balanced by the daily input of new pathogens. The longer the survival time in soil, the greater the equilibrium level.

TABLE 3-3. MAJOR ORGANISMS OF HEALTH CONCERN THAT MAY BE PRESENT IN SEWAGE FROM U.S. COMMUNITIES. (Akin et al., 1978)

Organisms	Disease	Reservoir(s)
I. BACTERIA		
Salmonellae (Approx. 1700 types)	Typhoid fever Salmonellosis	Man, domestic and wild animals and birds
Shigellae (4 spp.)	Shigellosis (Bacillary dysentery)	Man
Escherichia coli (enteropathogenic types)	Gastroenteritis	Man, domestic animals
II. ENTERIC VIRUSES		
Enteroviruses (67 types)	Gastroenteritis, heart anomalies, meningitis, others	Man, possibly lower animals
Rotavirus	Gastroenteritis	Man, domestic animals
Parvovirus-like agents (at least 2 types)	Gastroenteritis	Man
Hepatitis A virus	Infectious hepatitis	Man, other primates
Adenoviruses (31 types)	Respiratory disease, conjunctivitis, others	Man
III. PROTOZOA		
Balantidium coli	Balantidiasis	Man, swine
Entamoeba histolytica	Amebiasis	Man
Giardia lamblia	Giardiasis	Man, domestic and wild animals?
IV. HELMINTHS		
Nematodes (roundworms) *Ascaris lumbricoides*	Ascariasis	Man, swine?
Ancylostoma duodenale	Ancylostomiasis	Man
Necator americanus	Necatoriasis	Man

TABLE 3-3. MAJOR ORGANISMS OF HEALTH CONCERN THAT MAY BE PRESENT IN SEWAGE FROM U.S. COMMUNITIES. (Akin et al., 1978) (CONT.)

Organisms	Disease	Reservoir(s)
Ancylostoma braziliense (cat hookworm)	Cutaneous larva migrans	Cat
Ancylostoma caninum (dog hookworm)	Cutaneous larva migrans	Dog
Enterobius vermicularis (pinworm)	Enterobiasis	Man
Strongyloides stercoralis (threadworm)	Strongyloidiasis	Man, dog
Toxocara cati (cat roundworm)	Visceral larva migrans	Carnivores
Toxocara canis (dog roundworm)	Visceral larva migrans	Carnivores
Trichuris trichiura (whipworm)	Trichuriasis	Man
Cestodes (tapeworms)		
Taenia saginata (beef tapeworm)	Taeniasis	Man
Taenia solium (pork tapeworm)	Taeniasis	Man
Humenolepis nana (dwarf tapeworm)	Taeniasis	Man, rat
Echinococcus granulosus (dog tapeworm)	Unilocular echinococcosis	Dog
Echinococcus multilocularis	Alveolar hydatid disease	Dog, other carnivores

TABLE 3-4. PARASITES FOUND IN SLUDGE SAMPLES FROM 27 MUNICIPAL PLANTS IN SOUTHERN UNITED STATES. (Reimers et al., 1981)

Parasite Found	Probable Identity	Definitive Host
Ascaris eggs	*Ascaris lumbricoides*[1] *Ascaris suum*[1]	Humans Pigs
Toxocara eggs	*Toxocara canis*[2] *Toxocara cati*[2]	Dogs Cats
Trichuris trichiura	*Trichuris trichiura* *Trichuris suis*[3]	Humans Pigs
Trichuris vulpis eggs	*Trichuris vulpis*	Dogs
Toxascaris-like eggs	*Toxascaris leonina*	Dogs, cats
Ascaridia-like eggs	*Ascaridia galli* *Heterakis gallinae*	Domestic poultry Domestic poultry
Trichosomoides-like eggs	*Trichosomoides crassicauda* *Anatrichosoma buccalis*	Rats Opossums
Cruzia-like eggs	*Cruzia americana*	Opossums
Capillaria spp. eggs (3 or more types)	*Capillaria hepatica* *Capillaria gastrica* *Capillaria* spp. *Capillaria* spp. *Capillaria* spp.	Rats Rats Domestic poultry Wild birds Wild mammals (opossums, raccoons, etc.)
Hymenolepsis diminuta eggs	*Hymenolepsis diminuta*	Rats
Hymenolepsis nana eggs	*Hymenolepis nana*	Humans, rodents
Hymenolepsis sp. eggs	*Hymnenolepis* spp. (possibly more than one species)	Domestic and/or wild birds
Taenia sp. eggs	*Taenia saginata*[4] *Taenia pisiformis*[4] *Hydratigera taeniaeformis*[4]	Humans Cats Dogs

TABLE 3-4. PARASITES FOUND IN SLUDGE SAMPLES FROM 27 MUNICIPAL PLANTS IN SOUTHERN UNITED STATES. (Reimers et al., 1981) (CONT.)

Parasite Found	Probable Identity	Definitive Host
Acanthocephalan eggs	*Macracanthorhynchus hirudinaceus*	Pigs
Entamoeba coli-like eggs	*Entamoeba coli*[5] *Entamoeba* spp.	Humans Rodents, etc.
Giardia cysts	*Giardia lamblia* *Giardia* spp.	Humans Dogs, cats, mammals
Coccidia oocysts	*Isospora* spp. *Eimeria* spp.	Dogs, cats Domestic and wild birds, mammals

---

[1] Eggs of *A. Lumbricoides* and *A. suum* are indistinguishable.
[2] *Toxocara* eggs were probably mostly *T. canis*.
[3] *T.suis* eggs were probably only rarely seen.
[4] Eggs of these worms are indistinguishable.
[5] An intestinal amoeba that is a commensal, not a parasite.

TABLE 3-5. REPORTED PATHOGEN SURVIVAL TIMES. (Doran et al., 1977)

Organism	Survival Time	
	Plant Surface	Soil
Bacteria		
Salmonella	1-42 days	7-168 days
Shigella	1-7 days	
Mycobacterium	10-49 days	3-15 months
Leptospira		15-43 days
Erysipelothrix		21 days
Viruses		
Enterovirus	4 days	25-170 days
Poliovirus		32 days
Parasites		
Entamoeba histolytica	3 days	8 days
Ascaris lumbricoides ova	27-35 days	2-6 years

Sewage sludges contain a variety of mineral elements, such as Cu, Zn, Ni, Cd, Pb, Co, Hg, and Cr, which, when present in significant quantities, may be toxic to plants and animals, as well as a source of groundwater pollution. Generally, they tend to be absorbed strongly in soil. However, when the cation exchange capacity of the soil becomes exhausted due to exposure to more heavy metal ions than can be absorbed, the ions remain in solution and move with the groundwater. The tightly bound metals can also be transported to surface water and sediment by erosion of soil or sludge particles. Furthermore, solubilization of metals can occur as a result of biological reactions.

In their review of literature concerning potential groundwater pollution by trace elements in sludge-treated soils, Page and Chang (1975) concluded that, except for boron, movement of these elements in soil is restricted. However, significant downward movements of zinc, mercury, chromium, and others in soils to a depth ranging from 30 cm to 150 cm have been reported (MacLean, 1973; Lund et al., 1974; Boswell, 1975). The time necessary for these metals to travel through the soil profile may be quite long (Lieber and Welsch, 1954; Johnson et al., 1974; Josephson, 1976). The fact that deterioration of groundwater quality may result due to trace elements contamination cannot be ignored. The increasing use of subsurface sludge injection may cause more leaching of these elements to groundwater. Upward flow from a water table or when groundwater tables are high would also increase their pollution potentials.

The toxicity of many elements, such as hydrogen, lead, mercury, arsenic, sulfur, selenium, and tellurium, can also be enhanced by microbial methylation. This process would undoubtedly be intensified by land spreading of sludge, which increases both the microbial activity and the availability of these metals.

Although little information is available with respect to the phytotoxic substance formation from sludge applied to soil, the anaerobic decomposition of organic matter, resulting from excess application rates, will lead to the production of ammonia, hydrogen sulfide, and alcohols, all of which can be toxic to seed germination and plant growth. Volatilization of ammonia could also enhance eutrophication of surface water with readsorption.

Accumulation of nitrate in crops resulting from high concentrations of this plant nutrient in soil is well recognized. Plants also require small amounts of some trace metals, such as copper and zinc, for nutrition. These metals, together with others which have no known useful function, can also accumulate in plant tissues when their soil contents are high. Of the various trace metals, copper, zinc, and especially cadmium are of primary concern in most municipal sludges applied to agricultural land and forest due to their well-known plant and/or animal toxicities.

A recent review of literature concerning the effects of land application of sewage sludge on the cadmium and zinc content of crops (Council for Agricultural Science and Technology, Ames, Iowa, 1981) has concluded that plant uptake of these trace elements from soils is affected by the inherent differences among plant species, varieties within species, and different tissues within the same variety. Generally, Cd and Zn concentrations are greater in leafy vegetables and the vegetative parts of crops than in fruit, grain, or tubers. Among the soil properties, pH is the most critical factor in controlling plant uptake of Cd and Zn. Their concentrations in plants generally decrease with an increase in soil pH. Calcareous soils also minimize their uptake by crops. Nevertheless, the Cd concentration in crops grown on sludge-treated soil are generally elevated. It has been suggested that a Cd:Zn ratio of less than 1% in the applied waste should be maintained to generate protection of the ultimate food chain (Chaney, 1974).

One of the objectives of applying sewage sludge to land is to increase crop production and to substitute plant nutrients contained in the residues for those that would otherwise be supplied by commercial fertilizers. Phytotoxicity from compounds produced under anaerobic conditions and from Zn, Cu, and Ni resulting in crop failure or diminished yield is clearly undesirable. A greater concern is the accumulation of those toxic chemicals by plants that subsequently move to the human food chain, leading to potential human long-term toxicity. Presently, evidence on heavy metals toxicity through the food chain is focused on a relatively few elements such as cadmium, lead, and mercury. Because plant translocation of mercury and lead is limited, these elements do not pose as great a problem as cadmium.

Animals fed on sludge-grown forage may accumulate high concentrations of cadmium in their tissues, especially the liver and the kidneys. Bioaccumulation/biomagnification of this heavy metal not only adversely affects the health of animals, it can also be transferred to the human body.

Cadmium has no known biological function. However, after it is absorbed, the human body has no mechanism to excrete it. This trace element interferes with the actions of essential divalent ions, especially zinc, by displacing them from the functional sites. Bioaccumulation of cadmium has been associated with hypertension, deleterious effects on testicular function, and renal damage.

Plant accumulation of nitrates or nitrites and certain other trace elements has also been reported to affect animal health, ranging from loss of animal products and production to death. Careful analysis of sludge-grown forage crops, allowing for the supplement of macronutrients and diluting with nonsludge fertilized crops to reduce levels of micronutrients and heavy metals, may be necessary to protect animal and human health.

The presence of halogenated hydrocarbons (e.g., PCBs, DDT) and polycyclic aromatic hydrocarbons in some sewage sludges is also of concern with the land-spreading alternative (Dacre, 1980).

The environmental and health impacts of land application of sludge depend primarily upon the biological, chemical, and physical properties of both the sludge and the soil. Alteration of the organic/inorganic chemicals in sludge by the complex, dynamic soil systems may lessen or increase their environmental hazard. Many of these reactions are interrelated and/or interdependent. Because of these unique characteristics of each municipal sludge, differences in the soil properties and the environmental parameters, it is difficult to generalize as to whether or not a sludge can be used successfully for crop fertilization. Evaluation must be made locally and each case should be judged by its own merit. Special attention should be paid to the potential impacts of sludge application on groundwater quality deterioration and on bioaccumulation of toxic chemicals in crops. Its short- and long-term effects on animal and human health must be carefully considered.

3.2.2 Land Spreading of Industrial Wastes

Ross and Phung (1980) have summarized information related to land spreading (landfarming) of industrial wastes and the following is taken from their review. Industrial wastes that have been disposed of by this method are primarily from food processing, oil refinery, paper and pulp, tannery, and pharmaceutical industries. The wastes are composed mainly of organic material and are thus biodegradable. This practice is limited and will likely remain limited in applicability to about 3% of all industrial wastes. The landfarming alternative has been in use for 15 to 20 years by the petroleum industry for nonchlorinated organic waste.

Ross and Phung (1980) note that data on environmental contamination from landfarming of industrial wastes are scarce. However, it is reasonable to assume that some of the environmental concerns regarding landspreading of sewage sludge also apply to industrial wastes.

3.3 INCINERATION

Incineration is used in the disposal of sewage sludge and certain organic industrial wastes. We discuss first the incineration of sewage sludge and then examine some additional aspects associated with incineration of hazardous organic industrial wastes.

3.3.1 Incineration of Sewage Sludge

The major advantage of incineration is that it can destroy the potential biological pathogens and the organic matter present in sewage sludge, leaving an odorless ash. It may also eliminate certain hazardous chemicals or render them less toxic and/or harmless. Incineration generally reduces the total solid waste mass by about 90%. Thus, a great deal of problems associated with sewage sludge disposal is minimized. The current trend of fossil fuel price increase may also make the utilization of a heat recovery system that generates electricity or steam heat economically attractive.

The problems that may be associated with incineration are air and water pollution and the presence of heavy metals in the ash residues. These problems are accentuated when municipal solid waste is coincinerated with sewage sludge. It is generally agreed that carbon monoxide, hydrocarbons, oxygenated hydrocarbons, and other complex compounds can be produced due to poor combustion. Emission of some of these compounds is the source of odors. Inorganic gases such as sulfur oxides, ammonia, and halide gases are commonly produced from the sulfur, nitrogen, and halide content of the waste. A certain amount of nitrogen oxides, formed from the nitrogen constituents of the waste or from high temperature oxidation of nitrogen in the air, is also emitted. In the coincineration process, hydrogen chloride is of particular concern. There has been an increasing emission of this acid due to increased disposal of halidecontaining compounds such as polyvinyl chloride (PVC). In addition to its adverse health effects, it may cause corrosion of tube metal surface in steam-generating systems (Weinstein and Toro, 1976). Most of the inorganic gaseous emissions, however, can be significantly minimized by well-designed combustion chambers and careful control of operating conditions.

Of greater concern than the chemical gaseous emissions are the particulate emissions. Data showing particulate emission of a modern waterwall steam-generating incinerator upstream of the pollution control device are presented in Table 3-6 (Stabenow, 1972). Similar median values of particulate emissions (8.5 to 17.5 kg/metric ton of refuse) have been reported by various investigators (Chass and Rose, 1953; Duprey, 1968; Jens and Rehm, 1966; Niessen, 1970; Walker, 1964). Properties of particulates emitted from three conventional incinerator furnaces are shown in Table 3-7 (Walker and Schmitz, 1966). Particulate emissions can be significantly reduced by employing air pollution control devices such as electrostatic precipitators and various types of scrubbers. Certain percentages of the particulate matter, nevertheless, escape into the atmosphere. Deterioration of air quality is possible if incineration is widely employed. The "greenhouse" effect due to the release of overwhelming quantities of CO_2 resulting from combustion cannot be ignored.

It has been estimated that wastewater ranging from 2 to 12 tons per ton of solid waste processed may be generated in incinerator plants depending on the extent and mode of water uses in air pollution control equipment, residue conveying, and stack gas temperature control (Achinger and Daniels, 1970; Jens and Rehm, 1966; Weinstein and Toro, 1976). This process wastewater is contaminated by both dissolved and suspended materials (Table 3-8) (Achinger and Daniels, 1970) and is a potential source of pollution of streams and underground water.

Incinerator residues are classified into grate residue, grate siftings, and fly ash. Residues of sewage sludge incineration is mainly in the fly ash fraction. Significant quantities of the other two components are obtained if municipal solid waste is coincinerated, especially when front-end resource recovery is not practiced. These residues are commonly disposed in sanitary landfills. Although incineration destroys most of the organics and may render the inorganic salts less soluble (Ebhherhardt and Mayer, 1968), pollution of surface and groundwater by heavy metals due to landfilling disposal of these residues deserves our attention.

3.3.2 Incineration of Hazardous Organic Wastes

Disposal of hazardous wastes by high temperature oxidation or destruction is considered by many to be the ultimate disposal alternative. Temperatures ranging from 1000 to 3000°F can reduce hazardous liquid, solid, or gaseous waste to harmless compounds. Incineration of hazardous organic wastes can and has been carried out at sea. Incineration at sea may prove to be an environmentally, economically, and politically attractive alternative for waste disposal in the future (Halebsky, 1980).

One of the more versatile types of commercial incineration facilities found in the U.S. is a combination of liquid injection incineration and rotary kiln incineration. Spent solvents and other combustible liquids containing hydrocarbons or halogens are injected into a combustion chamber through an atomizing nozzle that mixes the waste liquids with air. Proper mixing with enough air allows complete waste combustion when kept at the elevated temperature for times approaching several seconds.

TABLE 3-6. PARTICULATE EMISSIONS FROM THE FURNACE OF A MODERN WATER-WALL INCINERATOR. (Stabenow, 1972)

Refuse charging rate, short tons/hr	16.6 - 16.7
Volume % CO_2 in flue gas (dry basis)	9.5 - 10.1
Dry catch particulates, lb/hr	379 - 427
Wet catch particulates, lb/hr	13 - 30
Total particulates, lb/hr	397 - 457
Dry catch particulates, lb/short ton	22.8 - 25.6
Wet catch particulates, lb/short ton	0.8 - 1.8
Total particulates, lb/short ton	23.9 - 25.2
Total particulates, kg/metric ton	12.0 - 13.7

TABLE 3-7. PROPERTIES OF PARTICULATES LEAVING FURNACES. (Walker and Schmitz, 1966)

Physical Analysis	Installation		
	(250 TPD)	(250 TPD)	(120 TPD)
Specific gravity, g/cc	2.65	2.70	3.77
Bulk density, g/cc (lb/CF)	--	0.495(30.9)	0.151(9.4)
Loss on ignition at 750°C, wt %	18.5	8.15	30.4
Size distribution (% by weight)			
2 microns	13.5	14.6	23.5
4 microns	16.0	19.2	30.0
6 microns	19.0	22.3	33.7
8 microns	21.0	24.8	36.3
10 microns	23.0	26.8	38.1
15 microns	25.0	31.1	42.1
20 microns	27.5	34.6	45.0
30 microns	30.0	40.4	50.0
Particulate emission rate, kg/MT (lb/ST)	6.1(12.1)	12.3(24.6)	4.6(9.1)

TPD = short tons per day
MT = metric ton
ST = short ton
CF = cubic foot

TABLE 3-8. TYPICAL WASTEWATER ANALYSES. (Achinger and Daniels, 1970)

	Quench Range	Scrubber Range	Final Effluent	
			Range	Average
pH	3.9-11.5	1.8-9.4	4.5-9.9	---
Temperature, °C	20-54	28-74	18-52	32
(°F)	(68-130)	(82-165)	(65-125)	(90)
Suspended solids, mg/l	140-1860	90-1350	40-580[a]	210
Dissolved solids, mg/l	360-2660	520-8840	320-4060	1190
Total solids, mg/l	610-3960	610-9160	610-4200	1400
Alkalinity, mg/l $CaCO_3$	90-720	0-80	15-310	135
Chlorides, mg/l	98-850	180-3540	95-1710	455
Hardness, mg/l	95-980	190-3430	100-480	240
Phosphates, mg/l	0.5-58	3-90	1-67	14

[a] After settling.

High Btu waste liquids are usually burned to provide heat to help operate a connected rotary kiln. The kiln is a cylindrical furnace, mounted horizontally at a slight incline, that turns slowly as heat is applied to the hazardous waste inside the unit. The rotary kiln can handle solids or semi-solids in 55-gallon drums. The drums are loaded into the kiln on a conveyor belt delivery system. Gases generated in the kiln are heated even further in a secondary chamber or afterburner, that also increases residence time. High Btu liquid wastes are normally incinerated in the afterburner during operation of the kiln to help attain the 2000 to 3000°F operating temperatures without using extensive amounts of auxiliary fuel. The solid residue or ash produced in the kiln is usually disposed of in a chemical landfill. The U.S. EPA has received a petition from an incineration facility to have its kiln ash delisted as a hazardous waste. This ash apparently passes all of the federal hazardous waste criteria and presents no concern for land disposal.

Combustion gases pass through several pollution control units before being emitted to the atmosphere. Particulates are removed by venturi scrubbers or electrostatic precipitators. Incinerators handling chlorine-containing waste generate gases containing hydrochloric acid (HCl) that are removed in a chemical scrubber by contacting with an alkali solution such as lime or sodium hydroxide. The alkali scrubber is purged as the HCl is neutralized. The purge stream or scrubber "bottoms" contains neutral salts of the alkali. This stream is usually treated and closely monitored before discharge to a surface water or publicly owned sewage treatment works. Parameters of concern include heavy metals and temperature.

Incinerator stack gases also contain carbon dioxide, nitrogen oxides, sulfur dioxide, water vapor, and traces of organics. Measured destruction efficiencies for organics are stated to be at 99.999+% removal.

The U.S. EPA has estimated that more than half of all hazardous waste could be safely disposed of via incineration. Only 6% of all hazardous waste has been disposed of by incineration because the landfilling alternative is expensive.

3.4 SOLIDIFICATION

Solidification is a detoxification process that physically stabilizes inorganic sludges by reducing the leaching rate of hazardous constituents such as heavy metals. There are five specific solidification techniques:

- cement-based, which incorporates the waste material into a cement-like material that can be disposed of in a landfill;
- lime-based, in which lime is mixed with the waste to increase the pH and therefore decreases the mobility of heavy metals;
- thermoplastic, in which the waste is encapsulated in a lightweight, sponge-like plastic that can be disposed of in a landfill;
- organic polymer, in which wet or dry wastes are blended with various resins such as urea-formaldehyde, resulting in solidification;
- glassification, which is used when the material is extremely dangerous or radioactive and the waste is combined with silica to form glass or a synthetic silicate material.

Solidification has been used to stabilize materials such as one-time pond cleanouts and other hazardous waste sludges. Most of these activities are done at the generator's site with solidified sludges disposed of at the same location. There is only one off-site commercial solidification operation in the eastern United States with its own disposal site. The stabilized sludges disposed of at this facility would pass the RCRA hazardous waste E.P. toxicity testing for leachate. The solidification disposal costs, however, approach those of regular land disposal in a chemical landfill, which has hindered acceptance of this disposal method.

Solidification of hazardous sludges creates a new waste material that can be land disposed without as great a concern about migration of metals into the environment. The solidification of organic sludges is not as successful in encapsulating the organic constituents. Compatibility testing is necessary to ensure that the sludge candidate for solidification can be successfully stabilized.

3.5 DEEP-WELL INJECTION

Deep-well disposal is the emplacement and storage of liquid wastes in the pores of deep subsurface geological formations that already contain unusable saltwater. Liquids potentially suitable for deep-well disposal include dilute or concentrated waste acids, weak or strong alkaline solutions, solutions of heavy metals, inorganic solutions, chlorinated hydrocarbons, various organic wastes, and solvents.

Amstutz (1980) has reviewed various aspects of deep-well disposal. He reports that deep-well disposal should be considered only when the liquid wastes cannot be treated or disposed of economically in other ways. The fundamentals of deep-well disposal are those of geology and the nature of the waste. There are areas in 36 of the 48 contiguous states which are underlain by sedimentary strata geologically attractive for deep-well injection.

The primary risk (other than waste handling and transportation) associated with deep-well disposal is the potential contamination of usable aquifers as a result of improper disposal or poorly designed deep-well injection systems. Satisfactory disposal in most but not all cases is limited to liquids that are free of suspended sediments or that can be filtered to provide a suitable filtrate (Amstutz, 1980).

4. ENVIRONMENTAL AND PUBLIC HEALTH IMPLICATIONS OF OCEAN DUMPING

This section of the report summarizes information on the fate and effects of ocean dumped materials. The intent of this section is to provide an overview of what is known regarding fates and effects rather than an exhaustive review of the literature. The objective of this overview is to aid in identifying gaps in our information regarding the fate and effects of materials dumped at sea in order to provide a basis for making recommendations for future research. These recommendations are provided in Section 5 of this report.

This section of the report is organized into three parts. Dispersion processes for ocean dumped wastes are summarized in subsection 4.1; general effects of ocean dumped wastes are reviewed in subsection 4.2; and a more detailed review of fate and effects of specific classes of contaminants (e.g., chlorinated hydrocarbons, radionuclides) is presented in subsection 4.3. We have chosen to focus on particular classes of contaminants for several reasons. First, major concerns regarding long-term effects of ocean dumping arise from questions related to the persistence and biological fate and effects of particular contaminants within the wastes. It is therefore necessary to understand the fate and effects of these contaminants in order to better evaluate the fate and effects of the whole wastes. Second, many wastes have a number of contaminants in common (e.g., chlorinated hydrocarbons or heavy metals) and, therefore, by focusing on classes of contaminants, it is possible to formulate research recommendations that address one or more types of wastes. Third, some wastes (e.g., dredge material, sewage sludge) are highly variable and an evaluation of the effects of such wastes requires an understanding of the major contaminants of concern; research that focuses on these contaminants will be more generally applicable than research directed solely at a particular type of waste.

4.1 DISPERSION PROCESSES FOR DUMPING METHODS

The impacts of wastes dumped in the ocean will depend on where, when, and in what concentration the wastes arrive at areas of concern such as shellfish beds, fisheries, or beaches. Movement and dilution or dispersion of waste will depend on the methods of dumping, the characteristics of the waste, and on the regional character of the ocean.

Dumping may occur from ships either directly or in containers such as drums. The dumped materials will vary in their density, in their phase (liquid, solid or particulate), and in their solubility and chemical characteristics. The local characteristics of the ocean that will affect waste dispersion include water depth, density stratification of the ocean, turbulence and wave activity in the area, and immediate and long-term currents. The following subsections discuss in general terms the dispersion processes in the ocean for three types of releases: liquids from ships, sludges and dredge materials, and slow bottom releases as from drummed wastes.

4.1.1 Liquids from Ships

In this subsection, we consider the release of wastes in the form of chemicals dissolved in water from boats or barges proceeding along a course in the ocean. This is a common method of disposing of acidic or basic wastes from chemical processes. Waste may be carried to specified dump sites in boats or barges with tanks that may hold hundreds of thousands of gallons of liquid. Commonly, the material is released into the ocean while the transporting vessel is proceeding along some specified course or pattern in the dump area. Regulatory control of such releases can include limits on the nature and concentration of the discharged material, location of the dumping site, times during which releases are permitted, and the steaming pattern and discharge rate in both liters per kilometer and liters per hour.

In the immediate wake of the discharge vessel, the waste material will be present in a specified concentration of liters per linear kilometer determined by the pumping rate from the discharge boat. The material will be spread in the vertical and horizontal in a pattern determined by the so-called "near-field" processes which include the speed and turbulent wake of the vessel, the momentum and discharge jet characteristics of the pumping system, and the position of the discharge relative to the boat wake. However, these complicated local processes are usually not of significance, because within a short time (generally less than an hour) natural processes in the ocean will spread the discharged material into a pattern much larger than the initial discharge wake.

The discharged material will undergo both lateral (horizontal) and vertical diffusion. Both of these processes are very dependent on the particular conditions prevailing at the location and time of the discharge, and specific measurements are needed for precise and quantitative calculations. However, considerable work has been done in the study of these dispersions and some generalizations are available which are often adequate for preliminary estimates of dumping impacts. An overall summary of measurements of lateral diffusion has been presented by Okubo (1971). These results, summarized in Figure 4-1, indicate the typical plume dispersion as a function of the size of the plume. Although a considerable range of dispersion is found among the measurements considered, there is sufficient information to make preliminary estimates.

The particular dispersion coefficient effective during a dump will depend on local conditions. Conditions which will tend to increase dispersion are strong waves, strong currents, and strong current shears. Current shears, in turn, will be induced by the presence of strong winds or the presence of currents in shallow water or near coastlines or oceanic fronts. For regulatory purposes, it is common to be concerned about worst case conditions that may occur when dispersion is minimal. For measurements to represent worst case conditions, they should occur during calm and wave free periods. Dispersion measurements are obtained by direct measurements of the area occupied by the plume of dumped material as a function of time. The concentration of dumped material in the wake of a discharge vessel can be measured directly in the case of certain chemicals that have properties easily measurable with shipboard equipment. Alternatively, barge loads or ship loads of discharge material may be spiked

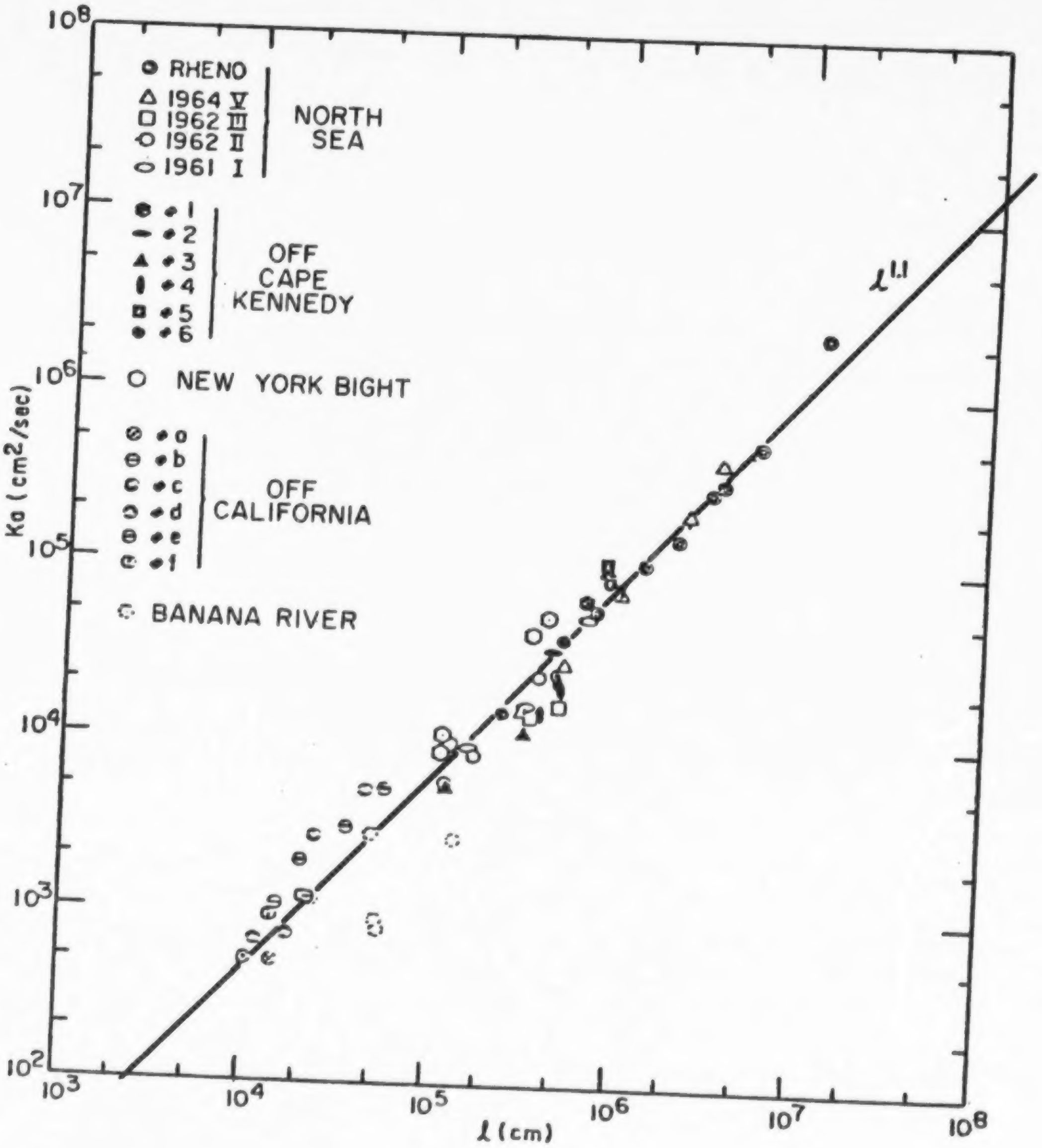


Figure 4-1. Lateral diffusivity K_a as a function of horizontal scale for a variety of areas and experimental conditions. (From Okubo 1971.)

with fluorescent dyes and the concentration of dye which represents the discharged material concentration can be measured with convenient, accurate, and sensitive fluorometers.

Vertical diffusion of discharged wastes is controlled by processes similar to those described above for lateral diffusion, and is also strongly affected by the density structure of the ocean and the relative density of the discharged material. In most cases, the discharge material is initially denser than seawater simply because it is most economical to concentrate the chemicals before barging them out to sea. This is advantageous from the point of view of dispersion, since discharges lighter than seawater would tend to pool on top of the ocean and be subject to considerably less dispersion as a function of time.

Discharges denser than seawater initially fall through the surrounding seawater in a turbulent process. As they fall, they entrain seawater and are diluted, and thus the density difference becomes smaller. This process clearly tends to be self-limiting. Discharges that are much denser than seawater are more turbulent and initially fall more rapidly, but also entrain seawater more rapidly. In any case, the discharged material gradually approaches the density of the surrounding water. The plume, now only slightly denser than surrounding water, sinks until it encounters a level in the ocean where the density of the ocean increases.

In most places in the ocean the first density increase, called a "pycnocline," is found at depths from 10 to 100 meters. The density above this pycnocline is frequently quite uniform as the ocean is stirred and mixed by surface winds. Thus, discharged material tends to rapidly diffuse down to the level of the pycnocline and then encounter denser water which may prevent it from sinking further. The most common form of vertical mixing, then, is for discharged material to be well mixed in the vertical from the surface down to the first major density discontinuity. This process generally takes a few hours.

Many variations on this basic pattern are possible. In shallow water where no pycnocline is present the waste will be mixed from surface to bottom. In the case of particularly dense waste with a strong shallow pycnocline, the waste material may fall down to the level of the pycnocline and not be found at shallower levels. In some cases the waste may still be dense enough at the time it reaches the pycnocline to fall through it into still deeper waters.

Worst case conditions for vertical dispersion will depend upon the particular biological communities of concern. If bottom communities such as shellfish are important, then well-mixed ocean conditions with no pycnocline present, which would allow wastes to fall rapidly to the bottom, could represent the worst case. In other circumstances, poorly mixed conditions, when winds are low and strong shallow pycnoclines are found as in the summer, would cause near-surface trapping of the waste which might represent worst case conditions for shallow planktonic communities.

As the discharge wake disperses under the processes described above, it is also advected away from the initial discharge region by the local currents. Ocean currents may be caused by tides, by wind driving, and by much more complicated large-scale ocean variability. It is important to remember that the actual current at the discharge time is going to advect the discharged wastes and that this actual current may be very poorly represented by an average or typical current. Variability of ocean currents is often quite pronounced and representation of the currents by a mean flow is rarely an accurate description. However, for regulatory purposes, it is often possible to generalize on upper bounds of local currents in order to determine worst case effects. Dispersion of discharged wastes generally reduces the concentrations to negligibly low values in the space of a few days time and circles can be drawn around the discharge region indicating the maximum extent to which currents may carry the significant concentrations of waste. Experimentally, in the study of discharges, it is common to place several drogues in the discharge wake of a barge in order to study the currents as they apply to a particular discharge. These drogues also have an experimental advantage in that they make it possible for a boat measuring the dispersion to keep track of the location of the discharge wake.

In general, it may be said that in the open coastal ocean, liquid wastes are diluted rapidly and their effects are local in time and space. However, this picture may be quite different in restricted bays or estuaries where long-term accumulation may be possible. In such cases, the history of the waste may have to be considered for time periods much longer than the day or two typical of the open ocean. Calculations of residence time of water in the restricted region may have to be made and consideration may have to be given to cumulative effects of many sequential dumps.

4.1.2 Sludges, Dredge Materials, and Precipitates

These wastes differ from the liquids described above in having a solids or particulates content as well as liquid. When discharged from ships, they behave in a manner bearing some similarities to the liquids considered above, but with differences that will be discussed here.

Initially these materials tend to be much denser than discharged liquids and, hence, tend to fall more rapidly through the water column. Their initial dilution is a matter of much greater complexity due to the variability that may be found in the behavior of the particulates content. The particles may have a wide range of sizes and they may also have varying tendencies to flocculate to form larger particles as they interact in physical and chemical processes with seawater.

Generally, discharges of such materials from ships sink in the ocean as an expanding cloud, which gradually increases in volume as it entrains surrounding water. It is important to note that the settling rate of this cloud may exceed the settling rate that would be measured for individual particles in laboratory experiments. This is because the aggregate collection of particles carries the water with it, and is less restrained by friction than individual particles moving through still water.

Some generalizations can be made. If the waste contains sufficiently large particles they will sink independently of the cloud in a rapid manner and proceed directly to the bottom. The remaining cloud will behave like a liquid but tend toward rapid settling characteristics. It may proceed directly to the bottom, passing through any intermediate pycnoclines, and then spread laterally on the bottom as a dense cloud. In the presence of a strong pycnocline, the descending cloud may be impeded at the density interface and spread horizontally while collapsing vertically into a thin "pancake" at the pycnocline. In any of these processes, there will be vertical separation of particles with different sizes and fall velocities. It should be noted that although these differing processes are intuitively comprehensible, they are much more difficult to model analytically or numerically than simple liquid discharges. A summary discussion of the calculations necessary to consider this kind of discharge is found in Koh and Chang (1973) and Brandsma and Divoky (1976).

A time series of seismic profiles shows a buildup of sludges in the New York Bight (NOAA, 1975). In the New York Bight, ocean dumping is the main source of modern sediments. The 30-foot (10-meter) accumulation of dredge spoil in approximately 33 years has resulted in the bull's eye pattern shown on Figure 4-2. This pattern indicates that the dumping activity has taken place at the properly designated site, and that transport of coarse sediments away from the site by natural processes did not keep up with the rate of dumping.

Some fraction of sufficiently fine particles will be left in suspension along with any liquid component of the waste. Since settling times for these fine particles may be days or weeks, their settling will be negligible compared with the dispersion processes acting on the liquids. They may be considered as simply another component of the liquid wastes, and their behavior may be calculated accordingly.

The behavior of a cloud of descending material that reaches the bottom rapidly will vary according to the bottom topography. On steep slopes, the cloud may flow downslope in the manner of a geological turbidity current. On flatter terrain, the cloud will be carried by the prevailing bottom currents and settle onto the bottom near to the discharge site. In the case of dredge material where the sediments are chemically and physically similar to the bottom materials already present, the discharge may be little noticed. However, other wastes with solid contents that differ chemically from the surrounding bottom may experience long-term chemical leaching.

Where pycnoclines or density fronts occur, finer particulate materials may accumulate, at least for short periods of time. This has been observed for sewage sludge, dredge material, acid iron waste, and drilling muds (Proni and Hansen, 1981; Orr, 1981a,b; Ayers et al., 1980; Pequegnat et al., 1978). This process and other processes affecting fate of discharged particulates in the water column are illustrated in Figure 4-3. As will be discussed later in this report, many contaminants (e.g., chlorinated hydrocarbons such as PCBs) tend to be associated with the finer particulate materials. Processes that enable these materials to accumulate and be available to biota are therefore important from the standpoint of addressing possible effects of these discharges. Further, density fronts are often areas of higher biological activity, and, therefore, there is a greater opportunity for organisms that congregate along these fronts to contact contaminants which may have accumulated there.

Contaminants associated with particulates accumulated along density surfaces pose special sampling problems. Acoustic techniques have been used to track these particulates for short periods of time (Proni and Hansen, 1981) and various sampling arrays, including multiport samplers, have been used to collect water samples. The problems are mainly defining the layers and then sampling these in some continuous or discrete fashion.

Much effort has been directed at measuring and estimating dispersion rates for materials within the water column. However, the fate of contaminants

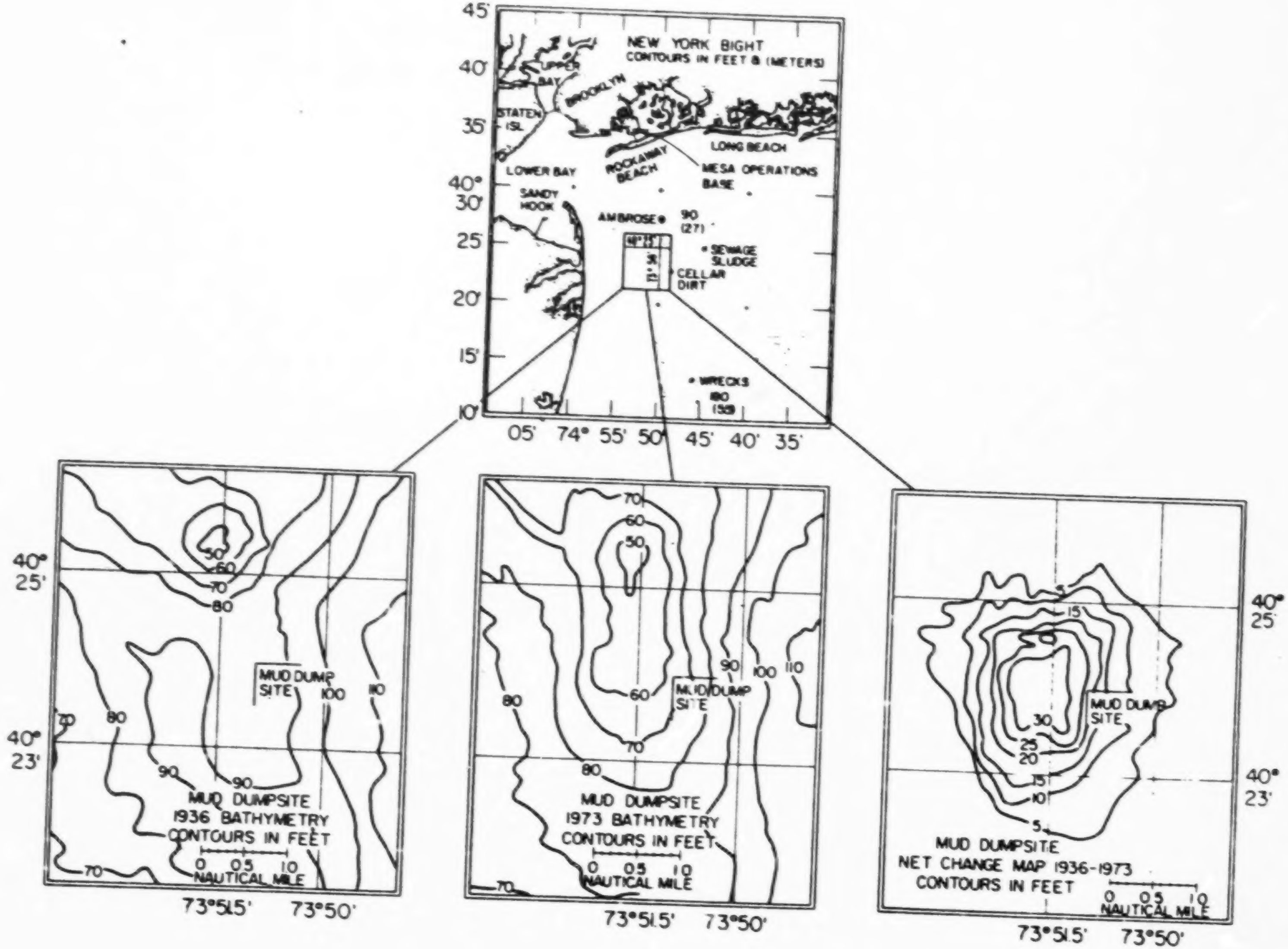

NEW YORK BIGHT
CONTOURS IN FEET & (METERS)
UPPER BAY
BROOKLYN
STATEN ISL
LOWER BAY
SANDY HOOK
ROCKAWAY BEACH
LONG BEACH
MESA OPERATIONS BASE
AMBROSE
90 (27)
SEWAGE SLUDGE
CELLAR DIRT
WRECKS
180 (55)
45'
40'
35'
40° 30'
25'
20'
15'
10'
05'
74°
55'
50'
45'
40'
35'
MUD DUMP SITE
MUD DUMPSITE
1936 BATHYMETRY
CONTOURS IN FEET
0 0.5 1.0
NAUTICAL MILE
MUD DUMPSITE
1973 BATHYMETRY
CONTOURS IN FEET
MUD DUMPSITE
NET CHANGE MAP 1936-1973
CONTOURS IN FEET
50
60
70
80
90
100
110
5
10
15
20
25
30
40° 25'
40° 23'
73°51.5'
73°50'

Figure 4-2. Bathymetric change at the dredge spoil (mud) dump site (NOAA, 1975)

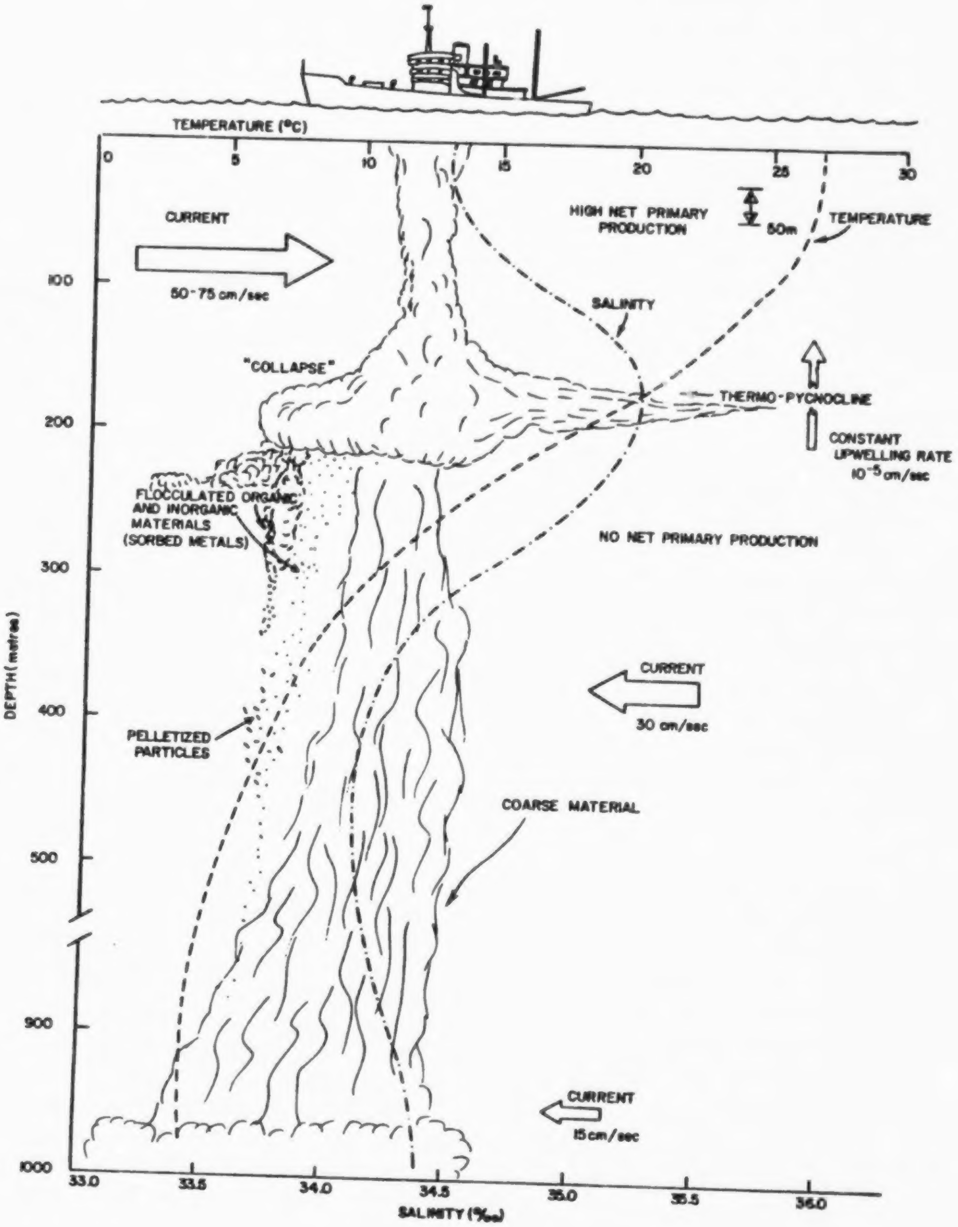

TEMPERATURE (°C)
0
5
10
15
20
25
30
CURRENT
50-75 cm/sec
HIGH NET PRIMARY PRODUCTION
50m
TEMPERATURE
SALINITY
"COLLAPSE"
THERMO-PYCNOCLINE
CONSTANT UPWELLING RATE 10^{-5} cm/sec
FLOCCULATED ORGANIC AND INORGANIC MATERIALS (SORBED METALS)
NO NET PRIMARY PRODUCTION
CURRENT
30 cm/sec
PELLETIZED PARTICLES
COARSE MATERIAL
CURRENT
15 cm/sec
DEPTH (metres)
100
200
300
400
500
900
1000
33.0
33.5
34.0
34.5
35.0
35.5
36.0
SALINITY (‰)

Figure 4-3. A schematic illustration of the physical fate of solids in wastes disposed of at sea (Pequegnat et al, 1978).

associated with particulates on the seafloor is more difficult to assess and at present no dynamic description of the behavior of sediment-fluid interactions exists that can be used in a practical application of a computer model (Ansyferov, 1980). It has been suggested that suspended load during major storms or other energetic events is responsible for the greatest flux of sediment transport and that other modes of transport (bedload transport, bedform translation) can be ignored on the continental shelf (Smith, 1977). During a transport event, the surface sediment will be reworked to a scour depth; eroded sediment will be laterally displaced. The erosional process will tend to homogenize the surface sediments (owing to turbulent mixing of the suspension), the transport and depositional process will tend to grade the sediment. Quiescent depositional areas are regions where contaminants associated with finer particulates could accumulate; therefore, information on the location of such areas in relation to the disposal and subsequent transport and dilution of wastes is important in assessing possible long-term effects from their contaminants.

4.1.3 Slow Bottom Releases

This category includes any materials released at the bottom from slow physical or chemical dissolution processes such as eventual leakage from drummed waste, decomposition of rubble or granular waste, or chemical leaching from accumulating mounds of sludge disposal. (It does not consider releases from bottom discharge pipes.) Some materials, such as radionuclides which may leak from drums, will become adsorbed to sediments and their fate will depend on sediment dynamics and adsorption coefficients. This will be discussed in more detail later in this report. Clearly, the release rates for the varying processes are quite dependent on the particular mechanisms in action. For liquids dropped to the bottom in containers, for example, some casually designed containers could break on impact but Greek amphora have been recovered still containing wine after two thousand years on the bottom. We will not attempt to consider the release mechanisms here, but simply discuss where released materials from the bottom will be transported and dispersed in the ocean.

The transport of bottom-released materials may be divided into shallow and deep cases. The shallow cases include those regions where the water column is frequently well mixed with no pycnocline and wave action is an effective turbulence mechanism at the bottom. Depending upon local conditions, this shallow category may apply only to waters less than 10 meters deep or as much as a hundred meters deep. In shallow waters, as defined, the transport and dispersion of material released at the bottom will be essentially similar to the calculations for liquids released from ships as described above. Considering the slow discharge rates that will probably apply, the concentrations resulting from such processes may be expected to be quite low. However, it is important to bear in mind that these concentrations may be chronic and persist for sufficiently long times to cause effects that would not be found for isolated releases.

For slow bottom releases in deep water, the released material will be found in a region of the ocean known as the bottom boundary layer. This region is the near-bottom analog of the well stirred region found below the surface winds and represents the adjustment of interior currents in the ocean to the friction of the bottom. The bottom boundary layer may range up to 75 to 100 meters in thickness, though it is generally less than this. Water within the bottom boundary layer will be moving somewhat more slowly than the prevailing currents and will be turned by the Coriolis force.

In regions of intermediate depth (10 to 100 meters), conditions will vary throughout the year such that the region may sometimes behave as a shallow region and sometimes as a deep region. That is, the density structure and turbulence within the ocean will vary and releases from the bottom will similarly vary in their dispersion and direction of travel. Evaluation of chronic releases in such areas may require several years of field measurements for adequate statistical characterization.

Several peculiarities of bottom releases may occur in particular regions. In canyons extending offshore of river mouths through the continental shelf region, it is typically found that the prevailing bottom flow is directed upslope toward the river mouth. This result, somewhat contrary to expectations, is an extension of the typical estuarine circulation pattern. It tends to be a variable phenomenon, fluctuating with time, location, and the particular geometry of the canyon. There are also regions of the ocean noted for their general upwelling characteristics, where deeper and denser offshore water is brought to the surface by a combination of winds, prevailing currents and bathymetry. In such regions, material released near the bottom will tend to be brought back to the shore and surface, and obviously such regions should be avoided, where possible, for dumped waste.

4.2 GENERAL EFFECTS OF OCEAN DUMPING

The effects of ocean dumping may be considered in terms of the nature of effect (physical, chemical, biological), duration of effect, and spatial extent. Much of the information on effects of ocean dumping of various wastes have been reviewed in several recent reports (National Academy of Sciences, 1976; NOAA, 1979a,b; NACOA, 1981; Ketchum et al., 1981). Many other reports on effects of ocean dumping are listed in the Reference section of this report.

The general effects of ocean dumping particular classes of wastes are presented in Table 4-1. Many of these effects are well known and have been studied extensively. Others either have posed problems from the standpoint of study or still remain as major concerns. In particular, the long-term fate and effects of persistent contaminants within the wastes appear to pose the major problems for study. For this reason we have in the following subsection of this report, focused on the fate and effects of major classes of contaminants. One of the general effects presented in Table 4-1 is substrate modification. Change in sediment characteristics can effect benthic larval recruitment. We mention this here because there is evidence that it occurs but that it has not been specifically addressed in past studies. Physical or chemical alterations of the sediments could render them less attractive to settling benthic larvae. Larvae of many benthic invertebrates are planktonic and, when ready to settle on the seafloor, have the ability to "test" the sediment and either settle (if appropriate factors are in order) or be carried further along in search of a more appropriate substrate (Chia and Rice, 1978). Such "avoidance behavior" could result in a reduction of population size of invertebrates for which the sediments have been rendered unattractive. This kind of sublethal effect cannot be evaluated by standard toxicity tests. Evidence for such effects is available from studies of drilling muds (Menzie et al., 1980; Tagatz et al., 1980) and possibly fly ash (based on our review of available field and laboratory studies which show low toxicity but also depauperate bottom areas).

TABLE 4-1. GENERAL EFFECTS OF OCEAN DUMPING VARIOUS KINDS OF WASTES

Effect	Waste Type					
	Sewage Sludge	Dredge Material	Coal Ash	Acid Waste	Fish Waste	LLW (Drum)
PHYSICAL						
• Burial	Limited	Yes	Yes	No	Limited	Minor
• Turbidity	Yes	Yes	Yes	Yes	Limited	No
• Reef effect	No	No	Yes (if blocks)	No	No	Yes (drums)
• Substrate modification	Yes	Yes	Yes	No	Limited	No
• Pycnocline accumulation	Yes	Yes	Yes	Yes (Iron Waste)	Limited	No
CHEMICAL						
• Changes in solubility of hydrocarbons	Increased DOC	Minor	No	Temporary	No	No
• Contaminants in water column	Temporary	Temporary	Temporary	Temporary	Temporary	No
• Contaminant levels in sediments	Yes	Yes	Yes	Limited	Limited	Yes (if leaks)
• pH changes	Temporary	No	No	Temporary	No	No
BIOLOGICAL						
• Biostimulation of phytoplankton	Yes	Yes	No	No	Yes	No
• Bioaccumulation	Metals Hydrocarbons	Metals Hydrocarbons	Metals	Metals	Limited	Radionuclides
• Benthic impacts	Yes	Yes	Yes	Limited	Yes	Limited
• Pathogens	Yes	Limited	No	No	No	No

The following subsection of this report considers the fate and effects of particular classes of chemicals discharged into the oceans.

4.3 FATE AND EFFECTS OF WASTE CONSTITUENTS

Various wastes that may be ocean dumped have a number of waste constituents in common (Table 4-2). These constituents vary among wastes as well as within a particular kind of waste. Sewage sludge and dredge material provide good examples of variable wastes. Such variability can be found in levels of metals, halogenated hydrocarbons, and polynuclear aromatic hydrocarbons. By focusing on particular constituents, it is possible to develop information that can be used to assess effects of ocean dumping of a number of wastes. It also facilitates identifying gaps in information or analytical techniques and in making appropriate research recommendations.

The major classes of waste constituents provided in Table 4-2 include those that may result in coastal eutrophication (nutrients) or that tend to be persistent, possibly resulting in long-term effects (e.g., halogenated hydrocarbons). Constituents that tend to be rapidly lost from the system upon discharge (e.g., volatile liquid hydrocarbons) are not considered; these constituents may be acutely toxic and such effects tend to be relatively well documented. Based on our review of available information and discussions with other researchers, it is evident that the fate and long-term effects of various waste constituents is of primary concern. An additional class of waste constituents which will be considered in this section of the report are radionuclides associated with low-level radioactive waste.

4.3.1 Nutrient/Organic Enrichment

Since the work of Ryther and Dunston (1971), the biostimulatory effects of nutrient enrichment upon marine systems has been clearly recognized. Well-documented examples of these effects have been summarized by Curl et al. (1979) for Chesapeake Bay, New York Bight, San Pedro Bay, San Francisco Bay, Kaneohe Bay, the Baltic, Aegean, and Adriatic Seas, Oslo Fjord, and several bays in Japan and Long Island, New York. These effects generally result from nutrient enrichment, particularly nitrogen. Since nitrogen apparently limits biomass production in marine environments, considerable effort has been expended by marine biologists to understand the nitrogen cycle. Recent reviews of nutrient cycling in coastal ecosystems (Nixon, 1981) and seawater (Fogg, 1982) summarize the present state of knowledge. An understanding of the processes involved in this cycle and the biological and geochemical agents responsible is essential to predictions involving the consequences of overloading the cycle directly by anthropogenic input or short circuiting it due to ocean disposal of materials harmful to the biological transformation processes. Inputs to the oceanic nitrogen cycle include nitrogen fixation, river flow, precipitation, and dry deposition. Precipitation is the largest single input. River input is the second major source and is growing apace with world industrialization. Walsh et al. (1981) estimated that the nitrate content of rivers draining developed areas has doubled in the last 25-50 years. Losses of nitrogen from oceanic environments occur during denitrification and loss of nitrous oxide. Denitrification is the major loss and both Fogg (1982) and Nixon (1981) stress its importance to the global nitrogen

TABLE 4-2. CHARACTERISTICS AND CONSTITUENTS OF SELECTED CLASSES OF WASTES WHICH MAY BE OCEAN DUMPED.

General Characteristics	Sewage Sludge	Dredge Materials	Fly Ash	FGD Sludge	Acid Iron Waste	Fish Waste
Density (g/cm^3)	0.58-1.2				1.08-1.15	
% Solids	5-10	20-80	98-99%	30-80	<1-2	28-38
pH	5.3-11.7		3-12	2.8-12.8	<0.1-2.2	
Nutrients			Negligible	Negligible		
Organic C (ppm)	65,000-480,000				3-3,600	
Total N (ppm)	1,000-176,000	200-4,000				14,000-23,000
NH$_4$-N (ppm)	1,000-67,600	100-2,000				
NO$_3$-N (ppm)	9-4,880					
Total P (ppm)	1,000-61,000	500-2,000				21,300-29,500
Total S (ppm)	6,000-15,000					
Metals						
Aluminum (ppm)	1,000-35,000					139-759
Arsenic (ppm)	6.0-230	0.1-585	3.2-74	4.0-12.0	<5-525	
Barium (ppm)	21-8,980		700-15,000	20-4,400		2.0-9.3
Beryllium (ppm)						
Boron (ppm)	4-757		179-1,040	42-211		6.0-14.0
Cadmium (ppm)	4-846	0.01-97.2	0.39-5.3	0.4-25	<0.025-900	
Chromium (ppm)	17-99,000	1.0-1,127	3.6-28.0	1.6-5.2	2-900	4.7-9.3
Copper (ppm)	84-10,400	0.05-5,680	43-238	39-104	0.12-8.17	7.5-13.4
Iron (ppm)	0-15,300	1,000-9,400			14,000-75,200	83-860
Lead (ppm)	13-19,730	<1-1,595	4.0-27	1.6-290	0.27-76	
Manganese (ppm)	18-7,100	24-4,680	157-374	56-340		<1.0-31.6
Magnesium (ppm)	300-19,700					1,700-2,200
Mercury (ppm)	1-10,000	0.006-4.77		0.01-0.46	<0.005-71	
Molybdenum (ppm)	5-39		5.9-12.0	8-81		
Nickel (ppm)	10-3,515	2.3-6,925	34-108	13-75	0.2-65	
Selenium (ppm)			1.7-16.4	2-4		
Silver (ppm)						
Titanium (ppm)					70-33,700	
Vanadium (ppm)		5-194	<25-<100	50-100		
Zinc (ppm)	101-27,800	0.009-15.5	92-386	14-2,050	5-530	61-88

TABLE 4-2. CHARACTERISTICS AND CONSTITUENTS OF SELECTED CLASSES OF WASTES WHICH MAY BE OCEAN DUMPED (CONT).

	Sewage Sludge	Dredge Materials	Fly Ash	FGD Sludge	Acid Iron Waste	Fish Waste
Halogenated Hydrocarbons	(Very Limited Information)		Unknown	Unknown	Expected to be Minor	
Polychlorinated Biphenyls (ppm)	0-352	0.033-7,762				
Dieldrin (ppm)	0.03-2.2	<0.008-0.088				
Aldrin (ppm)	0-16.2	<0.004-0.055				
DDTR (ppm)	0.1-1.1	<0.002-0.238				
Chlordane (ppm)	3.0-32.2					
	Many Other Compounds					
Polynuclear Aromatic Hydrocarbons	(Very Limited Information)		Unknown	Unknown	Expected to be Minor	Unknown
Benzo[a]pyrene (ppm)	0.01-1.7	<0.0003-15				
Benzo[e]pyrene (ppm)	0.02-1.4	0.059-1.000				
Benzo[b+k]fluoanthene (ppm)	1.0-5.78					
Dibenz[a,j]anthracene (ppm)	0.03-0.26					
Chrysene (ppm)	0.11-1.31	0.40-3.000				
Indeno(1,2,3,c,d)pyrene (ppm)	0.38-1.87					
Fluranthene	4.28	0.110-0.790				
Benzo(g,h,i)perylene (ppm)	0.02-1.10	0.66-0.280				
Pyrene (ppm)	0.2	0.50-7.200				

cycle. Regeneration and recycling of these nutrients occur in both the water column and in underlying sediments. As Nixon points out, however, it is not clear which major group of organisms is responsible for most of the organic matter decomposition and nutrient regeneration in coastal waters, although the role of microplankton and free-living bacteria is being examined (Fogg, 1982). A clear positive relationship has been established between primary production, organic input, and benthos consumption of organic matter (Nixon, 1981). Therefore, inputs of nitrogen to the marine system have implications not only for water column production but also for the benthos.

An example of relative contributions of nitrogen by ocean dumping to a local nitrogen budget is provided by the New York Bight (O'Connor et al., 1981). The total nitrogen input to the Bight from all sources is estimated to be 520 metric tons/day (MTD). Approximately 85 MTD (16%) is due to ocean dumping. Dredge material contributes 63 MTD of this while sewage sludge (17 MTD) and chemical waste (5 MTD) contribute the remainder. Curl et al. (1978) point out that ocean dumping has little effect on the eutrophication problems of the New York Bight. Urban discharge is a far greater contributor of nitrogen to the system.

Shiff and Morel (1979) have summarized the fate of dissolved nutrients and organic matter introduced to marine systems. Depending upon hydrographic conditions, introduced nutrients/organics will be diluted and advected upon disposal. As much as 90% of ammonia nitrogen will be volatilized from surface slicks but may be resolubilized over adjacent areas, thus increasing the areal distribution. Hydrolysis of dissolved organic matter may occur in the water column; however, nitrification, particularly of particulate material, occurs more frequently in the sediments (Bellen, 1978). Dissolved inorganic nitrogen (NH_4^+, NO_3, and NO_2) will be taken up by the phytoplankton, with ammonia taken up most rapidly. Maximum uptake rates for ammonia within sewage plumes may be high enough to remove the dissolved fraction on time scales of days. The particulate organic fraction may be transported out of an area or may settle to the benthos and enrich sediments. Under stratified conditions, fine particulates may accumulate on density surfaces.

The effects of the above enrichment extend from the water column to the benthos. Increased primary production will result from increased nutrients. This has clearly occurred in the New York Bight. Increase in production results in increased extracellular release of dissolved organic carbon (DOC) by the phytoplankton. This has apparently occurred in the New York Bight (Thences et al., 1979). Increased DOC results in increased solubility of alkanes and isoprenoid hydrocarbons as demonstrated in Narragansett Bay (Boehm and Quinn, 1979). Phytoplankton composition often changes under nutrient-rich conditions to nearly monospecific assemblages such as Nannochloris atomus in Moriches Bay (Ryther and Dunstan 1971) and the large mats of Microcystis aeroginosa often found in Chesapeake Bay (Curl et al., 1979). Marine Ecosystem Research Laboratory (MERL) tank experiments have demonstrated depressed zooplankton fecundity and abundance under nutrient enrichment probably as a consequence of various chemical changes resulting from increased primary production (C. Oviatt, pers. comm.). In Kaneohe Bay, Hawaii, shifts to microzooplankton have been observed (Curl et al., 1979) with increased primary production. Finally, organic enrichment of sediments may result in physical changes (grain size) or chemical changes that result in changes in community structure. Organically enriched sediments may serve as storage areas for metals, chlorinated hydrocarbons, or PAHs.

4.3.2 Metals

The fate and effects of metals in the marine environment and their public health implications have received considerable attention over the past decade. Much of the current literature has been reviewed by Förstner and Wittman (1981). As indicated in Table 4-2 on waste constituents, most of the wastes that may be ocean dumped contain metals, sometimes in relatively high concentrations. Ocean dumping can represent a significant input of metals to the marine environment. For example, the relative contributions of total metals due to dumping in the New York Bight have been estimated by Mueller et al. (1976) as follows: cadmium (82%), chromium (50%), copper (51%), iron (79%), mercury (9%), lead (44%), and zinc (29%) (Figure 4-4). While these appear to be relatively significant contributions, it is important to note that the effects of metals will depend not on the total amount dumped but on their availability to biota (Lee, 1977).

The fate of metals in the marine environment is dependent upon several factors including metal speciation, physical nature of receiving environment, physical nature of waste (liquid, sludge, solid), precipitation, complexation, sorption-desorption processes, and redox conditions of bottom sediments. The greater part of dissolved heavy metals transported by natural water systems is, under normal physicochemical conditions, rapidly adsorbed onto particulate material and eventually becomes part of the sediments. Thus, the fate of particulate and sedimentary material is an important factor determining the fate of metals in the marine environment. Studies supported by the U.S. Army Corps of Engineers, as part of their program to develop dredge materials disposal criteria, have shown that sediment-associated metals (i.e., Cu, Pb, Cd, Zn, Ni, Hg) are not readily released to the water column when contaminated sediments are resuspended in laboratory tests (Lee et al., 1975a,b, 1978; Lee and Mariani, 1977; Lopes, 1977). In general, these studies have shown that iron and manganese are the only metals released to the water column in large amounts from dredged sediments. All other metals tested were either sorbed, not released, or released in sufficiently small quantities to not significantly change the concentrations of metals at an open-water disposal site.

Förstner and Wittman (1981) have noted that heavy metals immobilized in bottom sediments can be released as a result of particular chemical changes. Remobilization is mainly caused by four types of changes in waters:

1. Elevated salt concentrations, whereby the alkali and alkaline earth cations can compete with metal ions sorbed onto solid particles.
2. Changes in the redox conditions, usually in conjunction with a decrease in the oxygen potential due to advanced eutrophication. Iron or manganese hydroxides are partly or completely dissolved, part of the heavy metal load being released.
3. Lowering of pH, which leads to dissolution of carbonates and hydroxides, as well as to increased desorption of metal cations due to competition with H^+ ions.

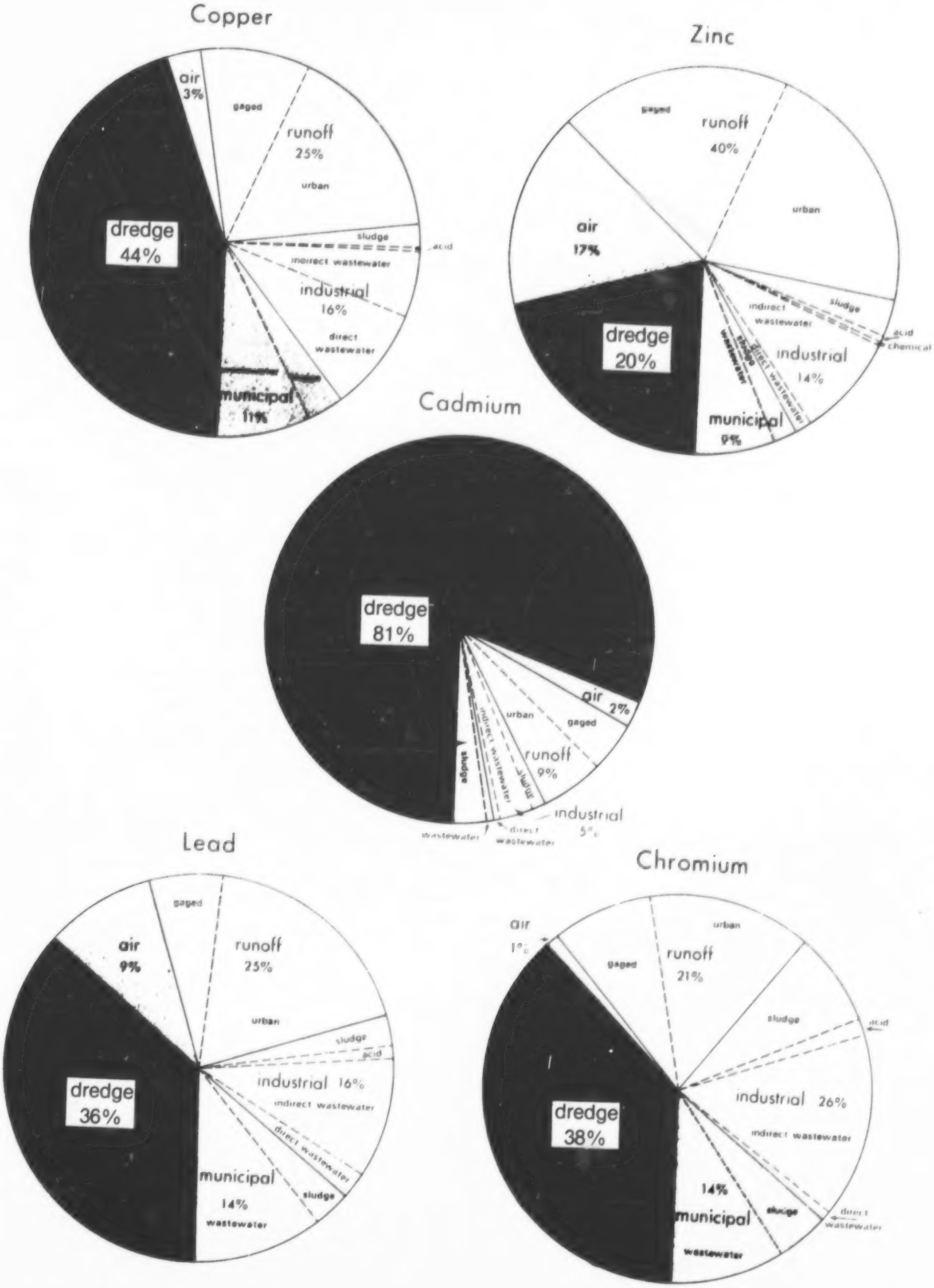


Figure 4-4. Relative contribution of various sources of metals to the New York Bight (Mueller et al., 1976).

4. Increased use of natural and synthetic complexing agents, which can form soluble metal complexes (sometimes of high stability) with heavy metals that are otherwise adsorbed to solid particles.

In addition to these four processes, there are other biochemical transformation processes by which heavy metals are transferred from the sediment to animal or plant organisms. Microbial activity can also play a role in mobilization of heavy metals.

There are three major microbial processes leading to mobilization of metals:

1. The destruction of organic matter to lower molecular weight compounds, which are more capable of complexing metal ions.

2. Changes in the physical properties of the environment by metabolic activities.

3. Conversion of inorganic compounds into metal complexation with organic substances, by means of oxidative and reductive processes. This final mechanism is exemplified by the alkylation of the elements mercury, arsenic, and selenium.

When uptake of toxic substances by microbiota occurs, these organisms are frequently able to "detoxify" these chemicals but, in the process, yield products that may be more toxic to higher organisms (Wood, 1974). This is the case for bacterial production of methyl mercury. The ability of microorganisms to detoxify their environment through organo-metallic transformation is not restricted to mercury, although the process is particularly important with regard to the widespread pollution by mercury. It has been established that mechanisms of biologic methylation are also effective in the formation of volatile compounds of As, Pb, and Se (Förstner and Wittman, 1981). The subject is also discussed in Craig (1980).

Recent reviews of the bioaccumulation of trace metals by marine organisms (Bryan, 1977, 1980; Cole, 1979; Förstner and Wittman, 1981) indicate that accumulation is largely governed by the speciation and bioavailability of the metals. Although dissolved metals in solution are generally the most available to organisms, metals in food organisms and particulate matter can also be important sources of metals to marine animals (Bryan, 1980). Bioaccumulation has been shown to be dependent on the form of the metal, its competition between chemically similar ions, environmental factors (e.g., temperature, salinity, D.O., pH, light), the state of the organisms, and the organisms' ability to regulate or metabolize the metal. Bioaccumulation by some marine species is directly proportional to external concentrations, while others have the ability to regulate uptake.

Biomagnification of potentially toxic metals up the food chain is an important consideration because of possible effects on man from eating contaminated seafood. Available evidence suggests that most metals are not biomagnified up the food chain (Bryan, 1979, 1980). A possible exception is methyl mercury, which does not appear to be regulated by any of the lower organisms or the majority of fish. However, it should be noted that, even for this metal, the high concentrations observed at higher trophic levels may not be attributable solely to food intake. In fact, water may be the most important source. In this context, the speciation of heavy metals in the water plays a role which is decisive not only for the acute toxicity but also for the bioavailability of the metals. The reason that the thesis of bioamplification of heavy metals along the food chain is not generally valid lies in the discrepancy between past, overly simplified investigations and more recent, better differentiated studies of food web enrichment processes (Förstner and Wittman, 1981).

Organisms differ in their ability to store and/or excrete metals that have been absorbed (Bryan, 1979, 1980). Storage may involve binding to proteins (e.g., metallothionein types), deposition in skeletal material, or intracellular storage. Losses to surrounding water occur via diffusion over general body surface, secretions, excretions, and molting. Toxic effects on organisms probably depend, in part, on the degree to which they can "handle" the metals taken up from the environment. Organisms have been shown to exhibit three types of metal contaminant regulation: the contaminant is excreted at a rate proportional to the body burden (e.g., Pb in Mytilus edulis); the contaminant is stored rather than excreted (e.g., Pb in Scorbicu laria plana), and organism excretes most of any excessive input (e.g., Zn in Carcinus maenas). Generally, fish and decapods are the best regulators, and essential metals such as Zn and Cu are better regulated than nonessential metals such as Cd and Hg.

Jenkins et al. (1981) have discussed the role of metallothioneins in sequestering and detoxifying metals in fish. These proteins have a high affinity for metals including Ag, Au, Cd, Cu, Hg, and Zn (Kagi and Nordberg, 1979). Synthesis of metallothionein is induced by low levels of Zn, Cu, Cd,and Hg, thus allowing organisms to respond to increasing quantities of trace metals by synthesizing this specific metal-binding protein (Brady et al., 1979; Bremer and Davis, 1975; Richards and Cousins, 1975).

Metallothionein is thought to exert its protective effect by sequestering free metal ions and partitioning them away from potential sites of toxic action (Jenkins et al., 1981). These authors note, however, that the ability of an organism to synthesize metallothionein may be exceeded so that additional metals are no longer sequestered. This saturation of the detoxification mechanism could result in spillover of excess trace metals and a concomitant onset of metal toxicity. Because of organisms' ability to detoxify trace metals, it has been suggested that the toxicological impact of trace metals may be evaluated most effectively by examining the relative distribution of trace metals between metallothioneins and potential sites of toxic action (Bayne et al., 1980; Engel and Fowler, 1979).

Acute toxic effects of metals in the marine environment are not likely except, perhaps, in the immediate vicinity of the discharge. However, sublethal and chronic lethal effects of metals on marine organisms may occur. Yet, there is little field evidence of such effects. Most observations of these effects have come from laboratory experiments (Bayne, 1979, 1980). Sublethal and chronic lethal effects of metal contamination are more likely to occur in benthic and epibenthic species because of elevated levels of metals in sediments. However, uptake of metals from sediments will depend on their availability. Combinations of metals may have synergistic or antagonistic effects (Förstner and Wittman, 1981). For example, mixtures of Cu and Zn or Cu and Cd produced more than additive effects in depressing embryonic development in sea urchins; Cu and Ni act additively. Synergisms or antagonisms may occur between metals and hydrocarbons. As already discussed, the toxic effect of metals taken up by organisms will depend, in part, on the ability of the organism to handle (e.g., detoxify or excrete) the metal.

An aspect of metal contamination which is not strictly "toxicity"-related is the possible effect on phytoplankton growth rates and species composition. Laboratory evidence suggests (as does a basic knowledge of factors affecting phytoplankton growth) that shifts in the composition of dissolved metals in the euphotic zone can alter the growth rates and species composition of phytoplankton (Murphy, 1981). Under realistic ocean dumping conditions, factors such as dilution and duration of exposure would be important in determining the extent to which such changes could occur.

With regard to effects on man it should be noted that with the exception of the Minamata experience for methyl mercury, there are few documented cases of human poisoning as a result of metals in seafood. Nevertheless, elevated levels of nonessential metals, such as Cd and Hg, are of some concern.

4.3.3 Polycyclic Aromatic Hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are composed of two or more fused aromatic (benzene) rings. Of primary environmental concern are the mobile compounds ranging in molecular weight from naphthalene to coronene. Two classes of PAHs have been delineated according to molecular weight: the lower molecular weight 2-3 ring aromatics and the higher molecular weight 4-7 ring aromatics. Compounds in the lower molecular weight class are acutely toxic to aquatic organisms, whereas those in the higher molecular weight class are not. However, all of the proven PAH carcinogens are in the higher molecular weight class.

It is estimated that nearly 700 metric tons of the PAH benzo[a]pyrene (BP) alone and 230,000 metric tons of total PAH enter the aquatic environment annually (Neff, 1979). Of this, domestic and industrial wastes account for an estimated 29 metric tons/year (MTY) (4%) of BP and 4400 MTY (2%) of total PAH. As indicated in the table of waste constituents (Table 4-2), sewage sludge and dredge material are two waste types that may contain PAHs and may be ocean dumped. Although ocean dumping of these wastes is not a major contributor of PAHs to the marine environment, it may be locally important.

Information on the fate and effects of PAHs in the marine environment has been reviewed by Neff (1979) and Stegeman (1981). PAHs entering the marine environment are rapidly adsorbed to organic and inorganic particulate materials and, as sedimentation occurs, they become associated with bottom sediments. There, PAHs may accumulate in high concentrations. Marine organisms often contain tissue PAH concentrations orders of magnitiude higher than aqueous concentrations, but equal to or less than those in bottom sediments.

Marine organisms are able to accumulate PAHs from water, food, and sediment. In most cases, accumulation from water is more efficient than from food or sediment. However, the widespread occurrence of PAHs in sediments suggests that bioavailability of sedimentary PAH is critical in determining exposure for animals, especially benthic and demersal species (Stegeman, 1981). Bioavailability is strongly influenced by association of PAHs with colloidal or particulate material. Sorption-desorption equilibria of PAHs with particulate matter clearly indicate that some fraction of PAHs is in solution in sediment pore water. Therefore, animals "compete" with sedimentary material for the available PAHs in solution. The dynamics of these competing processes determines, in part, the rate of PAH uptake.

PAHs are acutely toxic to marine animals at concentrations of about 0.2-10 ppm (in water) and deleterious sublethal responses are sometimes observed at concentrations in the 5-100 ppb range (Neff, 1979). The work of Hutchinson et al. (1979) has indicated that hydrocarbons with low vapor pressure and low water solubility (e.g., PAHs such as anthracene, phenanthrene, and pyrene) are more toxic on a molar basis than the high vapor pressure, highly water soluble compounds (e.g., benzene). These investigators postulate that the strongly lipophilic nature of hydrocarbons causes them to be rapidly and powerfully absorbed by lipid-containing cells and suggest that the lipoprotein cellular membrane may be a key site of action. However, the limited bioavailability of sediment-adsorbed PAHs would tend to render them substantially less toxic than aromatic hydrocarbons in solution. This underscores the importance of sorption-desorption processes in the sediments in affecting the availability of PAH.

Most animals are able to degrade PAHs to more polar metabolites and excrete them (Neff, 1979). Biotransformation of PAHs by marine animals appears to involve cytochrome P-450-dependent mixed-function oxidase systems. Species lacking PAH-metabolizing systems are able to release accumulated PAHs if placed in a PAH-free environment. Thus, food-chain biomagnification of PAHs occurs only to a very limited extent, if at all.

Some PAH metabolites are carcinogenic, mutagenic, or teratogenic. Rather than enhancing detoxification, metabolism of some carcinogenic PAHs in induced animals could result in a higher steady-state level of toxic products (Stegeman, 1981). Although studies with various carcinogens have demonstrated that chemicals can cause cancer in aquatic species, most attempts to demonstrate carcinogenesis by PAHs in aquatic species have produced equivocal results (Pliss and Khudoley, 1975). Although recently there has been some evidence that PAH can cause cancer in aquatic animals, there is to date no demonstration of PAH-induced carcinogenesis in marine vertebrate or invertebrate species (Neff, 1979; Stegeman, 1981). Recent studies in the Duwamish River and Hudson River estuaries have identified populations of Dover sole and Atlantic tomcod with very high incidences of hepatocellular carcinoma (McCain et al., 1977; Smith et al., 1979), and higher incidences of similar diseases have been reported for the Southern California and New York Bights. Although the etiology of such diseases in fish is uncertain, there is reason to suspect that the chemical environment is responsible, and PAHs have not been exonerated (Stegeman, 1981). The question of whether PAHs induce cancer in marine species, particularly at known ambient levels, is yet to be answered.

The importance of PAH occurrence in marine species as a source of these compounds to man is not well known. Neff (1979) concludes that PAH-contaminated water and fishery products represent minor sources of PAH toxicity for man. However, Stegeman (1981) notes that although PAHs in marine species, including bivalve molluscs, generally represent only a modest contribution to the total of these compounds in the human diet, there are occasional samples that can contribute substantial amounts of these carcinogens. Yet even these may be minor relative to other mutagens in the diet (Nebert et al., 1979).

4.3.4 Halogenated Hydrocarbons

The fate and effects of polychlorinated biphenyls (PCBs) and other halogenated hydrocarbons (Dieldrin, PBBs, DDT) in the marine environment is of concern because a number of these compounds are persistent (not easily degraded), can be accumulated in marine organisms, and can exert short- and long-term effects. Among the wastes which can be ocean dumped (Table 4-2), sewage sludge and dredge materials contain variable amounts of halogenated hydrocarbons, depending on regional inputs from industrial and municipal sources. Ocean disposal of halogenated/chlorinated hydrocarbon wastes or products is prohibited and these materials are generally disposed of by land-based alternatives or incineration at sea. The concern, with regard to ocean disposal, is for the fate and effects of these chemicals when they occur as contaminants in other wastes (e.g., sewage sludge). These sources can be important on a regional scale. For example, for the New York Bight it is estimated that sewage sludge contributes approximately 30% of total PCB loading and dredge materials contribute most of the remainder (Swanson et al., in press). However, the bioavailability of PCBs associated with sewage sludge is probably greater than that for dredge materials.

PCBs have received a great deal of attention because they have been widely used, have been produced in large amounts, are persistent, are widely distributed throughout the world's oceans and throughout all trophic levels, and may cause acute and long-term effects. Although PCB manufacture and use in the United States were halted, the existing "reservoirs" make PCBs a present and future environmental concern. Because a comparatively large amount of information has been generated for PCBs, much of this subsection on halogenated hydrocarbons will focus on this particular family of chemicals. Where appropriate, information based on studies of other halogenated hydrocarbons (e.g., DDT) will be presented. It is hoped that this subsection will provide a general background on what is known about the fate and effects of halogenated hydrocarbons. However, it is important to note that the various hydrocarbons differ in their relative solubilities, persistence, and effects. It is also important to point out that for many (if not most) of these chemicals there is little data on their concentrations in sources and within the marine environment.

Studies of the fate of PCBs in aquatic and marine systems have shown that a major portion of PCBs becomes associated with particulate matter and is deposited in sediments (Maurer et al., 1977; Murakami and Takeishi, 1976; Frank et al., 1977). PCB properties of low solubility, high specific gravity, and high affinity for particulate matter indicate that waterborne contaminant transport will be highly correlated with sediment transport phenomena (Frank et al., 1977; Wilson et al., 1978). Further, the attraction of PCBs to finer-grained sediments (Maurer, 1977) and preference to certain species of clay (Murakami et al., 1976) and organic matter (Stoffer et al., 1977) strongly suggests that the distribution of PCBs in marine or aquatic environments will be associated with fine-grained sediment transport (Gilbert et al., 1973). Higher concentrations of PCBs in marine environments will be concentrated in low-energy areas which promote deposition (Stoffer et al., 1977). Other chlorinated hydrocarbons (e.g., DDT, Kepone) have also been shown to have a high affinity for sedimentary material.

Once PCBs are assimilated in sediments, they are persistent and resistant to degradation. A year after an accidental point discharge of the PCB Arochlor 1254 to an estuary was eliminated, oysters had depurated PCBs until a low, steady concentration was attained, while concentrations in sediments, although low, were unchanged (Wilson et al., 1978). Veith (1972) found the same concentration of PCBs in sediments 10 miles from the discontinued point source over a year after the initial investigation. During this same study, some evidence was obtained that PCBs with a lower percentage of chlorination may be degraded in marine aquatic sediments by microbial metabolism.

As was previously discussed for PAHs, the bioavailability of sedimentary PCBs and other persistent halogenated hydrocarbons will be critical in determining exposure for animals, especially benthic and demersal species. The dynamics of sorption-desorption processes within the sedimentary environment will determine, in part, the rate of uptake.

PCBs have been shown to accumulate and have both acute and chronic effects in a variety of marine organisms at all trophic levels. In phytoplankton and zooplankton, PCBs decrease the frequency of cell division (Fisher, 1975), and inhibit cell growth, carbon fixation, chlorophyll production, oxygen consumption, and protein and nucleic acid synthesis (Ewald, 1976). PCBs initially associated with microparticulates are rapidly incorporated into marine diatoms (Iseki et al., 1981). The accumulation of PCBs by diatoms occurs where PCB concentrations in water are nondetectable indicating a degree of bioaccumulation. Moore and Harris (1972) demonstrated that the PCBs Arochlor 1242 and 1254 reduced radiocarbon uptake at concentrations in water between 1 and 2 ppb. Laboratory LC_{50} values for marine phytoplankton were 6.5 ppb while in a natural community LC_{50} increased to 15 ppb (Moore and Harris, 1972).

PCBs introduced to a controlled ecosystem (Iseki et al., 1981) decreased primary productivity by 30% and replaced the population of diatoms with microflagellates. This type of community shift has been observed in pollution-effect experiments using copper and mercury (Thomas and Seibert, 1977). The initial introduction of PCBs to the water column also increased settling velocities of particulate matter but did not increase gross sedimentation rates due to reduced primary productivity (Smayda, 1970). PCBs eliminated zooplankton from the water column of a 64 m^3 enclosure within 15 days after addition of concentrations equivalent to 50 ppb (Iseki et al., 1981).

The chronic effects of PCBs to invertebrates have been demonstrated using predominantly laboratory, closed-system experiments (Iseki et al., 1981). Concentrations that produce the toxic effects are not normally found in ambient seawater. Due to the low concentrations of PCBs in water, there are strong indications that the primary mode of contamination is affected by the organisms' respective food source (McLease et al., 1980). In-situ studies indicate a strong correlation between PCB-contaminated sediments and concentrations in benthic organisms (McDermott et al., 1976; Halter and Johnson, 1977).

Fiddler crabs (Uca pugilator) continuously exposed to the PCB Arochlor 1242 under laboratory conditions exhibited a depressed rate of molting. Upon transfer to unpolluted water, the molting rate increased by 7-12%. However, residual effects from the accumulated PCBs prohibited molting from returning to normal rates for the duration of the experiment (Fingerman and Fingerman, 1979). The effects of bioaccumulation of PCBs on lobsters are presently being sought (Farrington, 1981).

The biological effects of PCBs on fish have been reviewed by Califano (1981). He reports that PCB toxicity in fish can result in direct mortality and, without observable sublethal effects, accumulated PCB residues may affect populations through reproductive failures and alteration of growth and activity patterns in young fish. Sublethal effects such as modification of biochemical and physiological processes can also result from PCB intoxication.

Toxicity to fish has been found to vary depending upon the PCB compound, species studied, and age of the fish. In general, PCB toxicity decreases as the percent chlorine content of the PCB increases. Early life stages are more susceptible to toxic effects than later life stages. The mechanisms of toxic action of PCBs in fish are not known and may be different for the various chlorinated biphenyls (Califano, 1981). PCBs may be directly toxic to fish or may become toxic following metabolic alteration.

Enzyme assays have indicated activation of microsomal enzyme systems and ATPase inhibition in PCB-exposed fish. On the basis of cytochrome P-450, cytochrome b_5, and NADH-cytochrome reductase, induction of liver mixed function oxidase system occurred with acute exposure to PCBs (Lipsky et al., 1978). It is thought that PCBs can induce metabolism of other compounds such as PAHs via the mixed-function oxidase system. This could result in an increased production within the animal of reactive metabolites of PAHs, assuming that there was a sufficient source of available PAH.

Halogenated hydrocarbons such as PCBs and DDT are much more resistant to metabolism by marine organisms than are PAHs. Therefore, they may accumulate in tissues to a greater degree than PAHs and have a greater potential for food-chain biomagnification. However, much of the apparent effects of food chain biomagnification would still occur if residues were not transferred along food chains, since levels tend to be related to lipid pools of organisms, and top predators such as seals have large fat deposits. Trophic-level effect and age effect may apply to a greater degree in larger organisms not capable of direct uptake from water, whereas influence of partitioning from water to body lipids is more likely to apply to small organisms. Organisms containing PCBs in their tissues have been found to be able to depurate these chemicals to varying degrees when placed in a PCB-free environment.

Although relatively few studies have examined the metabolic fates of PCBs in fish, there are indications that some chlorobiphenyls are metabolized at slow rates (Califano, 1981). The ability to metabolize PCBs is apparently species specific and the liver is the main organ involved in metabolism and excretion of PCBs in fish. Petersen and Guiney (1979), in their literature review on PCB metabolism, report that green sunfish and goldfish rapidly metabolize trichlorobiphenyl whereas bullheads and trout metabolize it only very slowly or not at all. While PCB metabolism is indicated in some fish species (Hinz and Matsumura, 1977; Sanborn et al., 1975; Grugen et al., 1976), the identity of PCB metabolites is virtually unknown.

Brown et al.(1981) have examined the metabolic fates of DDT, DDE, and DDD in white croakers (Genyonemus lineatus) from Southern California. The authors noted that although a considerable number of studies have been done on the detoxification of trace metals in organisms exposed in their natural habitats (discussed in Jenkins et al., 1981), little has been done to elucidate the ability of these organisms to detoxify xenobiotic hydrocarbons.

Brown et al. (1981) briefly reviewed the biochemical metabolism of xenobiotic hydrocarbons and the following is taken from that review. Metabolism usually involves, first, the addition of oxygen to the hydrocarbon to form a highly reactive and toxic epoxide intermediate. This intermediate can attach to macromolecules (e.g., proteins, DNA) with concomitant toxic effects, or alternatively, can be further metabolized to water soluble, less toxic metabolites. It has been suggested that the occurrence of toxic effects (i.e., whether epoxide intermediates will react with cellular macromolecules) depends on the ratio of the rates of bioactivation processes to deactivation processes. If the rate of deactivation processes is less than that of bioactivation processes, then levels of toxic epoxide intermediates will increase, causing toxic effects. It should also be noted that the parent compounds (e.g., DDT, DDE, DDD) also may have direct toxic effects.

Brown et al. (1981) summarized the major steps in metabolism of DDT (Figure 4-5). The first, and most likely rate-limiting step is the removal of chlorine to form DDE or DDD. The next series of steps can result in conversion of these to epoxides (described as reactive toxic intermediates) which can alternatively be converted, either spontaneously or enzymatically, to a variety of water-soluble products which may be excreted. Results of the study on white croakers indicated that fish from a site with low levels of contaminants and also a highly contaminated site (Palos Verdes), have the ability to metabolize (detoxify) one or more of DDT, DDE, and DDD (Brown et al., 1981). The investigators reported an increase in amounts of lipids in both liver and muscle tissue of croakers from Palos Verdes and presumably these lipids provide a reservoir for deposition of hydrophobic hydrocarbons. Quraishi (1977) notes that it is essential that DDT, DDE, and DDD be partitioned into lipid pools since otherwise they could act as severe neurotoxins.

As in the case of PCBs, the ability of animals to metabolize DDT and its derivatives appears to be species specific. Although Brown et al. (1981) observed metabolism in white croakers (as revealed by the presence of metabolites), Addison et al. (1977) indicated that trout do not have the ability to metabolize either DDT or DDE.

The information presented above indicates that the fate and effects of halogenated hydrocarbons depend on a number of processes ranging from physical transport mechanisms to enzyme reactions. The major processes include sediment transport and deposition, sorption-desorption rates, uptake of available hydrocarbons, and subsequent storage, metabolism, and excretion. The rates of processes are important in determining effects of halogenated hydrocarbons on particular organisms. Absolute concentrations in the sediments, water column, or biota do not necessarily indicate which effects are or may be occurring. It is also important to note that species differ in their ability to store, metabolize, or excrete xenobiotics and, therefore, results generated for a few species, while instructive, should not be extrapolated to the marine community as a whole.

4.3.5 Low-Level Radioactive Waste (LLW)

Low-level radioactive wastes (LLW) that may be disposed at sea include those which have low radiation levels at the outside of the container (90% fraction) as well as those (10% fraction) which may require remote handling or shielding (Table 4-3). The United States currently does not dump LLW at sea. However,

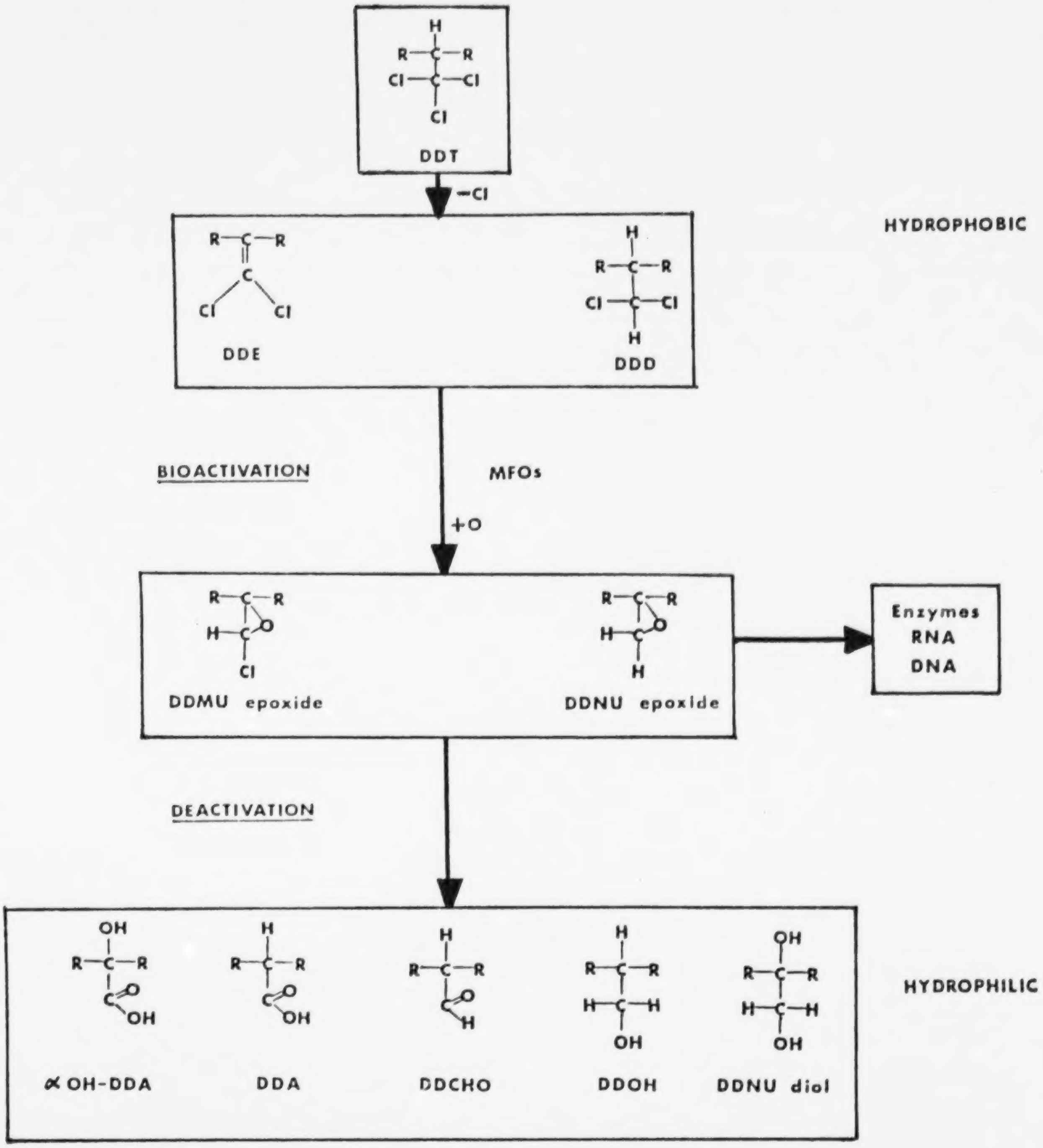

DDT
−Cl
HYDROPHOBIC
DDE
DDD
BIOACTIVATION
MFOs
+O
DDMU epoxide
DDNU epoxide
Enzymes
RNA
DNA
DEACTIVATION
αOH-DDA
DDA
DDCHO
DDOH
DDNU diol
HYDROPHILIC

Figure 4-5. A summary of the major steps in metabolism of DDT (Brown, et al., 1981).

TABLE 4-3. MAJOR RADIONUCLIDES IN LOW LEVEL RADIOACTIVE WASTE. (from MacBeth et al., 1979)

Nuclide	Halflife (yr)	Concentration in Low-Level Waste (90% Fraction) (Ci/m^3)	Concentration in Higher Radiation Level Waste (10% Fraction) (Ci/m^3)
^{3}H	12.3	0.12	
^{14}C	5730	3.8×10^{-3}	--
^{51}Cr	0.08	4.3×10^{-2}	65
^{54}Mn	0.86	2.5×10^{-2}	40
^{55}Fe	2.7	4.3×10^{-1}	--
^{58}Co	0.19	4.3×10^{-2}	65
^{59}Ni	8×10^{4}	1.3×10^{-3}	--
^{60}Co	5.27	0.13	200
^{63}Ni	100	0.24	--
^{90}Sr	29	4.8×10^{-3}	--
^{99}Tc	2.1×10^{5}	3.2×10^{-5}	--
^{125}Sb	2.73	5.3×10^{-3}	--
^{129}I	1.6×10^{7}	6.4×10^{-6}	--
^{134}Cs	2.06	4.8×10^{-2}	70
^{137}Cs	30.2	8.6×10^{-2}	130
^{152}Eu	13	4.8×10^{-5}	--
^{226}Ra	1600	1.2×10^{-4}	--
^{230}Th	7.7×10^{4}	7.1×10^{-5}	--
^{232}Th	1.4×10^{10}	8.4×10^{-7}	--
^{235}U	7.0×10^{8}	3.2×10^{-6}	--
^{237}Np	2.1×10^{6}	4.6×10^{-8}	--
^{238}U	4.5×10^{9}	7.1×10^{-5}	--
^{238}Pu	87.8	3.2×10^{-4}	--
^{239}Pu	2.4×10^{4}	4.3×10^{-5}	--
^{240}Pu	6540	6.7×10^{-5}	--
^{241}Pu	15	1.6×10^{-2}	--
^{242}Pu	3.9×10^{5}	2.4×10^{-7}	--
^{241}Am	433	3.0×10^{-5}	--
^{243}Am	7370	2.1×10^{-6}	--
^{243}Cm	28	6.0×10^{-7}	--
^{244}Cm	17.9	1.9×10^{-4}	--
	TOTALS	1.2	570

between 1946 and 1970 the U.S. dumped 60,000 curies of packaged LLW at coastal and offshore sites. Thirty-five sites were used, but 95% of the radioactivity was dumped at four sites: two off the Maryland-Delaware coast, one off San Francisco, and one off the Farallon Islands, Pacific Ocean (Dyer, 1981).

The fate of LLW in the sea includes radioisotope decay, sediment adsorption, transport due to erosion-deposition, release to the water column, biological uptake, and food-chain transfer. Table 4-3 presents the half lives of the common radionuclides in LLW. As indicated in subsection 3.1, 300 years are required to render the average LLW nonharmful. Release of radionuclides could occur if the containers in which LLW is packaged are breached.

Once released, radionuclides may be adsorbed to suspended particulates or sediments (IAEA, 1978). Concentration factors for radionuclides adsorbed to sediments are high for most radionuclides in LLW and generally range between 500 and 50,000 x that of water (NEA, 1980). The sediments of the seabed generally have strong sorption characteristics. The contaminated sediments may be transferred near bottom by transport of erosional sediments and particulates in the deep nepheloid layer (MacCave, 1978). Bioturbation may also effect sediment redistribution at dumpsites (Pneumo Dynamics Corp., 1961) and evidence exists that this may be the dominant mechanism by which at least cesium is redistributed (Dayal et al., 1979). These authors also conclude that migration of cesium via diffusion through pore water is insignificant. In general, they felt that the sediments are a barrier to release of cesium to the overlying water column (0.3% of cesium released from barrels enters water column).

Radionuclides can be bioaccumulated in the same manner as their cold isotopes. The return of radionuclides to the surface via the food chain pathway will depend, in part, upon the extent of benthopelagic coupling. The magnitude of such coupling and the degree to which radionuclides might be tranferred via marine food chains is unknown. However, fish associated primarily with the bottom have been caught far up in the water column (Haedrich and Henderson, 1977). A schematic illustrating possible fates of radionuclides in the marine environment is provided in Figure 4-6. As indicated, biological and physical routes of transport from deep-sea sediments to the upper water column are least known.

The dumping of LLW at sea may result in various effects. Radionuclides released from containers could be bioaccumulated by marine organisms and this may result in somatic or genetic damage. Increases in the radionuclide concentrations of invertebrates in the vicinity of the northeast Atlantic LLW dumpsite have been observed (Feldt et al., 1980) and radionuclides are known to accumulate in tissues of various marine organisms. The degree to which these accumulations would result in damage is presently unknown. However, at the implied maximum dose rates in both water and sediment at the northeast Atlantic site, the IAEA (1978) concluded that no significant deleterious effects would be expected on populations of marine organisms (including plankton). In the case of all the activity dumped over several decades remaining in the sediments of a very restricted area, the partial or complete destruction of the local benthic fauna could not be excluded. Regarding effects on mutation rates in populations of marine organisms, two IAEA expert panels have examined the effects of ionising radiation on aquatic organisms and ecosystems (IAEA, 1976). The evidence indicates that at the maximum doses inferred from estimated release rate limits,

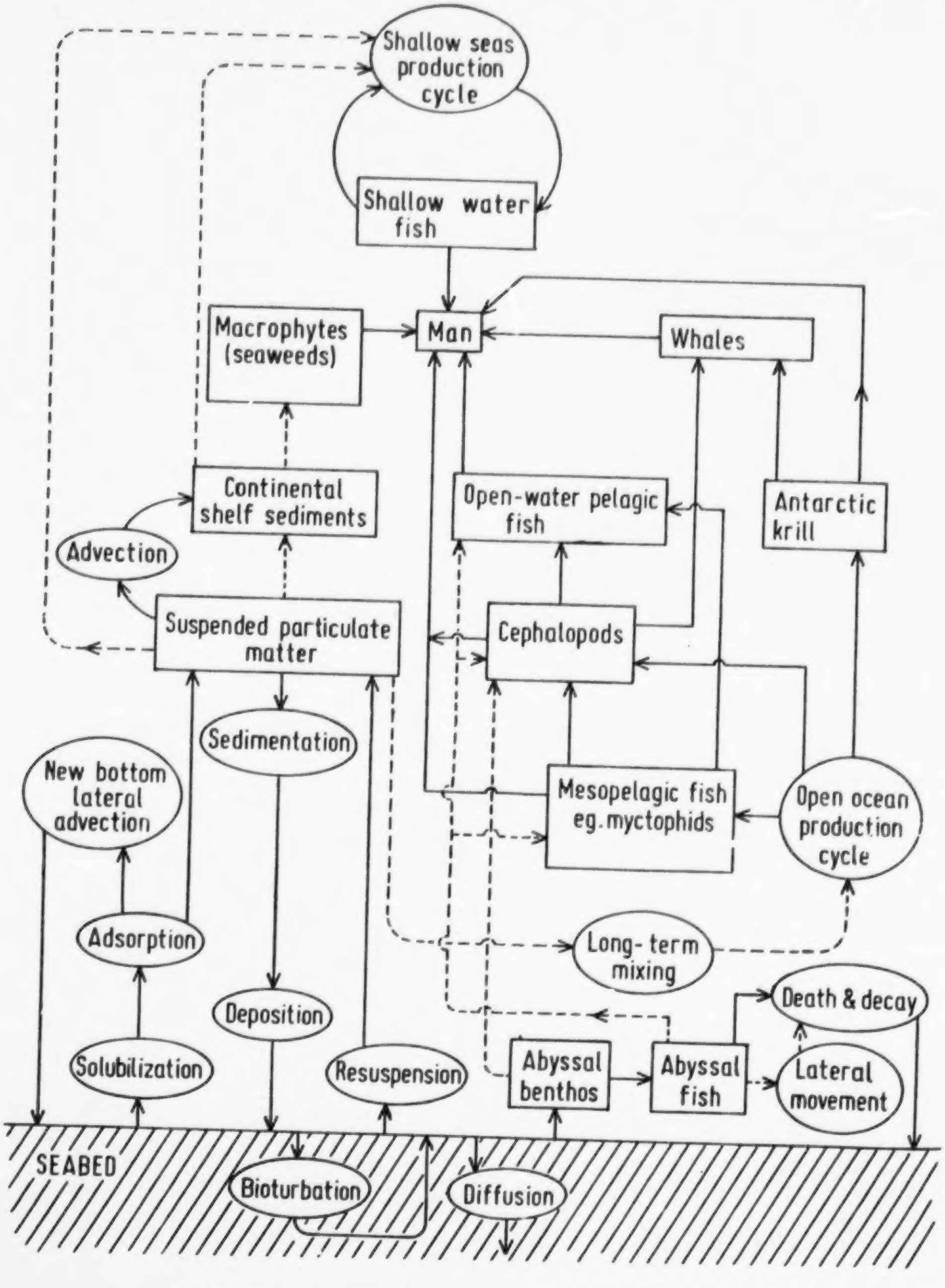

Shallow seas production cycle
Shallow water fish
Macrophytes (seaweeds)
Man
Whales
Continental shelf sediments
Open-water pelagic fish
Antarctic krill
Advection
Suspended particulate matter
Cephalopods
Sedimentation
New bottom lateral advection
Mesopelagic fish eg. myctophids
Open ocean production cycle
Adsorption
Long-term mixing
Deposition
Death & decay
Solubilization
Resuspension
Abyssal benthos
Abyssal fish
Lateral movement
SEABED
Bioturbation
Diffusion

Figure 4-6. Possible routes of radionuclide transport in the marine environment. Dotted lines indicate routes for which there is little information (from International Atomic Energy Agency).

no significant deleterious effects of a genetic nature are likely to be produced in populations of marine organisms.

Effects at the invertebrate community level have not been observed (Reish, 1977; Polloni and Williams, draft). Surveys of invertebrates at the Atlantic and Pacific dumpsites revealed no difference compared to control stations. Musick and Sulak (1978) came to a similar conclusion regarding the demersal fish community, but qualified their conclusions by noting that the mobility of the fish may have affected their results.

Increased biological production around the dumped cannisters due to "reef effects" also may occur. The strongest evidence of such increased production is the enrichment of nitrogen and organic carbon in a core taken from around a cannister at the Farallon Island site (Dayal et al., 1979). The authors suggest that although this finding may be due to a reef effect, the enrichment also may be due to paint that flaked off the cannisters.

Possible effects of deep ocean disposal of LLW on public health have been evaluated as part of a review of the continued suitability of the dumping site for radioactive waste in the northeast Atlantic (NEA, 1980). The evaluation utilized a generic approach developed by the IAEA. There are two parts to the IAEA assessment — an oceanographic model which derives water concentrations of radionuclides related to release rates and a radiological component which takes the water concentrations and evaluates the doses to man and the ecosystem which may result. The NEA (1980) note that ideally a site-specific model is required which correctly represents the oceanographic and biological processes in the area in question. Unfortunately, the understanding of these processes, being very limited at present, is not adequate to permit a site-specific model to be constructed in a satisfactory manner. The NEA report stated that eventually the necessary knowledge must be acquired. The first priority is considered to be continued investigations aimed at identifying critical pathways.

The rates of dumping of wastes at the northeast Atlantic site are summarized in Table 4-4, where they are also compared with the IAEA release rate limits. The IAEA figure for alpha-active wastes is based on long-term processes, which only become limiting for the dominant radionuclide Pu-239 after a long period of time (about 4000 years). Because wastes have been dumped for less than 30 years, the short-term rate limit which for Pu-239 is a factor of ten larger should be used, thereby reducing the percentage for overall alpha-active wastes to 0.08% (NEA, 1980).

TABLE 4-4. TOTAL AMOUNTS OF LLW DUMPED AT THE NORTHEAST ATLANTIC SITE AND CORRESPONDING IAEA RELEASE RATE LIMITS. (NEA, 1980)

Group	Total Amounts Dumped (Assuming no Decay Took Place)	Maximum Dumping Rate in Any One Year	Average Dumping Rate	IAEA Release Rate Limits(a)	% of IAEA Release Rate Limit
	Ci	Ci/y	Ci/y	Ci/y	
Alpha activity	8.3×10^3	1.4×10^3	750	10^5	0.8
Alpha activity (Ra-group)	100(b)	10(b)	10(b)	10^4	0.1
Beta gamma activity (excl. tritium)	2.5×10^5	4.3×10^4	3.6×10^4(c)	10^7	0.3
Tritium	2.6×10^5	1×10^5	4.3×10^4(d)	10^{11}	<<0.1

(a) INFCIRC/205/Add. 1/Rev. 1

(b) Estimate, no detailed information available over all years

(c) Average over 1975-1979

(d) Average over 1974-1979

5. RECOMMENDATIONS FOR FUTURE RESEARCH

The purpose of this section is to provide recommendations on future research needs regarding environmental fate and effects of wastes dumped at sea. These recommendations are based on our review of available information and our assessment of gaps in that information. Naturally, some research needs are greater than others either because there is a dearth of information on a particular subject or because the potential risks associated with ocean disposal of a particular waste require close inspection. We have attempted to take this into account and have assigned priorities to the various categories of study discussed in this section of the report.

During preparation of this report, the five-year marine pollution monitoring plan (Interagency Committee on Ocean Pollution Research, Development and Monitoring, 1981) was received. Its recommendations are generally similar and consistent with our assessment. That report's section dealing with research related to ocean waste disposal is presented here as Appendix A.

Our recommendations are more specific and presented within a different framework. Suggested categories of studies and their relative priorities with regard to various classes of wastes are presented in Table 5-1. A matrix format was chosen because research needs related to ocean dumping are often common to a number of waste categories. Priorities are ranked high (H), moderate (M), and low (L). Priority assignment is based on a number of factors including the extent of information already available and the relative environmental and public health risks associated with various classes of chemical contaminants. The priority levels provided in Table 5-1 are based on our assessment of the above factors, from literature reviews, discussions among the scientists who prepared this report, and discussions with a number of investigators at various laboratories.

Research needs can also be considered in terms of the probability that a particular waste will be ocean dumped and the comparative quantities of wastes. Based on information given in Section 2 of the report, the likelihood that wastes presented in Table 5-1 will be ocean dumped is as follows: sewage sludge (moderate-high), dredge materials (high), fly ash (moderate), FGD sludge (moderate), gypsum (low-moderate), acid iron waste (moderate-high), industrial sludges (low-moderate), low-level radioactive waste (moderate).

5.1 PHYSICAL FATE OF DISSOLVED CONSTITUENTS

Information on the physical fate of dissolved constituents (e.g., liquids discharged from ships) was discussed in subsection 4.1.1. The basic information required to assess physical fates include the scale of mixing and dispersion at

TABLE 5-1. CATEGORIES OF STUDIES AND THEIR RELATIVE PRIORITIES WITH REGARD TO OCEAN DUMPING OF VARIOUS WASTE TYPES. PRIORITY LEVELS ARE HIGH (H), MODERATE (M), AND LOW (L). SEE TEXT FOR FURTHER DISCUSSION.

Categories of Studies	Sewage Sludge	Dredged Materials	Fly Ash	FGD Sludge	Gypsum	Acid Iron Waste	Industrial Sludges	LLW
A. Physical fate of dissolved constituents	M	M	L	L	L	M	M	L
B. Physical fate of particulate materials	H	H	H	H	H	M	H	H
C. Effects of nutrient enrichment	H	M	L	L	L	L	L	L
D. Uptake, storage, depuration of metals	M	M	M	M	M	M	H	M
E. Uptake, storage, depuration of PAHs	M	M	L	L	L	L	L	L
F. Uptake, storage, depuration of halogenated hydrocarbons	H	H	L	L	L	L	L	L
G. Uptake, storage, transport of radionuclides								
H. Effects of sediment alterations on larval settlement	M	H	H	H	M	L	H	L
I. Toxicity studies (lethal/sublethal)	M	M	L	L	L	M	M	L
J. Site monitoring	H	H	H	H	H	H	H	H
K. Inventory of wastes constituents	H	H	L	L	M	M	H	L
L. Cross-media risk assessment	H	H	H	H	H	H	H	H

a dumpsite and within adjacent areas to which the waste would be transported. These data are provided by standard physical oceanographic measurement programs and are usually obtained as part of ocean dumping site selection and monitoring programs. We believe that this information is important in assessing fate of dissolved materials and measurements need to be made at new as well as existing sites. However, because there are ongoing programs providing this information, and because mixing and advection processes tend to be relatively straightforward, this category of study has been assigned a low to moderate priority status.

Design of a field program to determine appropriate physical oceanography parameters can be outlined as follows:

1. Parameters to be measured

The choice of parameters to be measured include: density stratification, determined by hydrographic (temperature, salinity) profiles taken from ships or by moored temperature-salinity recorders; current patterns, determined with moored current meters or by drogue tracking; direct dispersion coefficients, by survey of the concentration of discharged waste or dye tracer; and dispersion influences, by measurement of waves, current shears, and meteorological parameters such as surface winds.

2. Choice of time and duration of measurements

Measurements for dumping activities planned for months or years of continuing operations will have to provide a characterization of the statistical variation of oceanic parameters, and thus will have to span appropriate meteorological seasons, tidal variations, and fluctuations in the significant ocean conditions such as waves, fronts, eddies, and hydrographic seasons.

Measurements for individual or infrequent dumping procedures can be briefer attempts to represent expected "worst-case" conditions, such as calm conditions when dispersion will be minimal, or other conditions appropriate to the type of dumping.

Monitoring measurements of ongoing dumping activities are of value to verify predicted behavior of discharges and their impacts. Scheduling of such measurements is obviously limited by the schedule of the dumping activities. It is important to realize that, because of the oceanic variability, such measurements must be considered for their monitoring value rather than as predictive examples. Predictive value could result only from the monitoring and analysis of numerous individual dumping exercises.

3. Determination of spatial extent of measurements

The extent of the region impacted by dumped materials will vary with the planned volume of material to be dumped and repetition rate of dumping activity, the toxicity of the material and its tendency for neutralization through chemical or biological processes, the transport and dispersion processes that are present at the site, and the biological, economic, or political sensitivity of adjacent areas that may be affected. Field program planning must therefore be preceded by preliminary estimates of the impacted area based on the proposed dumping activity, the best available estimates of the concentrations of pollutants that will be of concern, dispersion estimates, best available estimates of currents in the region, and considerations of possible sensitive areas. If initial estimates show that the affected area may be of comparable size to a partially enclosed proposed dumping region such as a bay, gulf, or strait, then studies of that entire region may be required.

4. Consideration of special topics

Special conditions that may require particular field studies include: oceanic fronts in the vicinity of the proposed site; estuarine circulation patterns which may extend out to the site; or bottom boundary layer flows, sediment resuspension processes, and turbidity currents where dumped material may come to rest on the bottom.

5.2 PHYSICAL FATE OF PARTICULATE MATERIALS

This category of studies has been assigned high priority status because: 1) many of the more persistent contaminants in the marine environment (i.e., metals, PAHs, halogenated hydrocarbons, radionuclides) tend to become associated with particulate material; and 2) our ability to adequately sample and predict the fate of particulates in the water column and sediments is limited (see subsection 4.1.2).

Key areas where additional information is needed include the following:

1) The fate of discharged particulates in the vicinity of density fronts. Of particular interest is the degree to which particulates will accumulate along density surfaces, both in terms of duration and concentration. Multiport water sampling systems combined with acoustic techniques have been shown to be helpful in monitoring the fate of particulates. We recommend that the development and application of such systems continue. A research program should be carried out to provide adequate ground truth for acoustic measurements under a variety of conditions.

2) The fate of particulate materials when they reach the seafloor and become part of the sediments. Although much attention has been given to the dispersion and settling of discharge plumes within the water column, comparatively little information is available for predicting the fate of this material once it has reached the seafloor. However, the short- and long-term fate of this material is important with regard to possible effects on the marine environment. We recommend that an appropriate sediment transport model be developed to aid in evaluating the fate of contaminated sediments. The model(s) should be developed for major dump sites and would use appropriate physical oceanographic, geologic, and meteorologic data for the regions of interest. Alternatively, generic models may be employed where a number of sites have common conditions.

Additional data required for developing and verifying the model(s) should be provided by two basic kinds of studies. Laboratory sediment flume studies should be used to provide information on shear stress required to resuspend various sedimentary waste types (e.g., sludges, dredge materials) associated with particular sediment types. Other conditions which should be considered in these laboratory studies include various micro- and macrobiological factors that would tend to stabilize the sediments or which could be important in physical movement of contaminated sediment within the upper sediment layers (bioturbation).

The second study that is needed involves field verification measurements. These include well designed field monitoring programs to measure the spatial extent of contaminants associated with sediments at and around dumpsites. Many past programs have emphasized collecting replicate samples at several stations. However, it is our opinion that sampling should be spatially extensive and that sampling effort would be better allocated by increasing the number of stations at and around dumpsites rather than the number of replicate samples at a station. Prior to the onset of future sampling programs, the design of such programs should be examined by statisticians with expertise in the areas of sampling design and allocation of sampling effort. Their recommendations should be evaluated and incorporated as appropriate into the program.

An evaluation should be made of various sediment sampling techniques (grabs, cores, box cores, etc.) in order to establish which techniques are most suitable for obtaining undisturbed sediment samples. Contaminants associated with sediments should not be analyzed in entire large volume (e.g., 10 cm long) cores because of dilution effects. Analyses should be conducted on discrete layers of sediments. At the sediment surfaces, analyses should be restricted to the upper few millimeters; analyses should be conducted at 1-3 cm intervals in lower sediment layers.

Tripod camera/current meter systems should be employed at a few major dumpsites in order to establish physical oceanographic (and, to some extent, biological) conditions under which resuspension of sedimentary material occurs. These systems are capable of providing month-long records of bottom conditions and probably constitute the most useful field verification technique. We recommend that tripods be established at a few major dredged material dumpsites, and at the existing and proposed sewage sludge dumpsites.

Two other monitoring systems appear to be valuable as field verification techniques. These include camera systems designed to obtain still photographs or video data from the sediment surface and sediment profiles, and high frequency (high resolution) side-scan systems. The camera systems are capable of providing detailed "point sampling" data in an efficient and quick manner; the side-scan systems provide larger-scale information on the nature of the sediment surface. Together, these two "remote sampling methods" could be an effective means for rapidly monitoring dumpsites. We recommend that the application of these techniques be further investigated.

5.3 EFFECTS OF NUTRIENT ENRICHMENT

This category of studies has been assigned a high priority status for sewage sludge, moderate for dredged material and low for other waste types. This reflects the relative nutrient contributions of these waste types.

A considerable number of studies have investigated the effects of nutrient levels on phytoplankton growth rates and species composition. Processes related to nutrient regeneration are continuing to be studied. The fate of dissolved nutrients and the degree to which "excessive" levels are reached depends strongly upon physical oceanographic processes (dispersion, mixing, advection) and the studies recommended in subsection 5.1 of this report are directed at providing that information.

There is evidence that elevated nutrient levels can affect the health of the zooplankton and the concentrations of other chemicals in the water column (see subsection 4.3.1). In particular, MERL experiments in progress have shown that elevated nutrient levels can reduce zooplankton fecundity; the causal agent(s) have not been identified. We recommend that mesocosm experiments (like MERL) continue, to provide additional insights into the effects of elevated nutrient concentrations. We further recommend that an evaluation be made of the extent to which the MERL system can represent offshore (as opposed to purely estuarine) conditions, for the purpose of examining the effects of elevated nutrient levels (derived from sewage sludge) on offshore plankton communities.

5.4 UPTAKE, STORAGE, AND DEPURATION OF METALS

This cateogry of studies was assigned a primarily moderate priority status (high for industrial sludges) because: 1) there have been a considerable number of laboratory studies on bioaccumulation and depuration of metals; 2) with the possible exception of methyl mercury, there is little evidence that metals are biomagnified up the food chain; and 3) there are few documented cases, based on field studies, of metals causing environmental damage to marine organisms or public health problems (the Minamata incident is a notable exception.) On the other hand, metals are a class of contaminants found in almost all wastes that may be ocean dumped (see Table 4-2) and in some cases ocean dumping can represent a significant source of metals. Based on our review, we recommend studies in the following areas.

1) Studies (see subsection 4.3.2) have shown that marine organisms vary in their ability to regulate the uptake, storage, and excretion of metals. One mechanism which has been described as important with regard to an organism's ability to "detoxify" accumulated metals involves the protein metallothionein. It has been suggested that an organism's ability to sequester or otherwise detoxify or eliminate metals can be exceeded. This might occur if the amount of bioavailable metals increases, if the rate of bioaccumulation increases, or if an organism's ability to detoxify or eliminate metals decreases. We recommend that studies be conducted on various organisms' abilities to physiologically regulate accumulated metals.

Two basic kinds of studies are recommended. First, laboratory/mesocosm (e.g., MERL) studies using realistic exposure conditions would provide needed information on the uptake and subsequent fate of metals in marine organisms. Analyses of metals should be carried out for specific tissues or organs. The objectives of these studies would be to determine the uptake rates and internal concentrations that exceed the organisms' abilities to sequester, detoxify, or excrete the accumulated metals. These experiments should be carried out on a range of species.

Second, careful analyses should be made of metal concentrations in organism collected in the field. The objective of these field studies should be to evaluate the extent to which levels of accumulated metals have reached or exceeded the organisms natural storage or excretion systems. These patterns in organism concentration should be related to information on the concentrations of available metals.

2. The rate at which metals associated with sediments will become available for biological uptake will depend on adsorption-desorption rates as well as other chemical factors (see subsection 4.3.2). Additional information is required on these geochemical processes.

3. Items 1 and 2 above refer to rates and processes related to metal availability, uptake, metabolism, storage, and excretion. The aggregate of these processes will determine the effects that metals discharged into the marine environment will have on organisms. In order to assess or predict these effects, we recommend that a modeling approach which incorporates these various processes be developed.

4. Phytoplankton growth rates and species composition have been shown in the laboratory to be affected by changes in the concentrations of dissolved metals. Such effects could have important consequences at the ecosystem level, should chronic inputs of metals actually result in these changes. Effects will depend on the exposure, duration, and concentrations of various metals resulting from discharges. It is likely that at present rates of ocean dumping, such effects are insignificant. However, they should be considered more closely should frequency and volume of ocean dumping increase. Effects should be evaluated by means of laboratory growth rate studies as well as field data on baseline metal concentrations, and physical oceanographic conditions.

5.5 UPTAKE, STORAGE, AND DEPURATION OF PAHs

This category of studies was assigned a low or (for sewage sludge and dredged materials) moderate priority status because: 1) ocean dumping of most wastes either contribute little or no PAHs; in the case of sewage sludge and dredge materials, local inputs may be important; and 2) available evidence indicates that PAHs are metabolized or excreted rapidly by marine organisms and that there is little biomagnification up the food chain. The following studies are recommended.

1) The availability of PAHs associated with sediments to marine organisms will depend on adsorption-desorption rates. Organisms "compete" with particulate material for the available PAHs. Information on the rates of these processes and the ability of organisms to take up PAHs from sedimentary material is needed. These studies should be carried out in laboratory/mesocosm systems. Radiolabelled PAHs should be used in these experiments in order to quantify the rates of physicochemical processes as well as the rates of uptake by marine organisms.

2) Marine organisms have the ability to metabolize PAHs. However, some of the metabolites are reactive with concomitant toxic, mutagenic, or carcinogenic effects. The degree to which toxic effects will occur probably depends on the rates of uptake, metabolism (i.e., activation), and deactivation to more polar metabolites. Laboratory studies using radiolabelled PAHs should be used to evaluate these rates. Of particular interest are differences in the rates of activation and deactivation.

3) There is little or no information relating diseases in fish to environmental concentrations of PAHs. However, an ongoing effort should be made to relate histopathologic and other conditions in fish to concentrations of PAHs, halogenated hydrocarbons, metals, and other contaminants. Part of the problem of establishing cause and effect relationships is that there are a number of potential causal agents, and effects are not necessarily tied to any one of these. From a statistical standpoint, this is a multifactoral problem. We recommend that existing or proposed field sampling programs be carefully evaluated by statisticians with expertise in experimental design. Their recommendations should be incorporated into ongoing programs.

4) As the recommendations presented above suggest, effects of PAH on the marine environment are the result of a number of processes and the rates at which they occur. Information presently existing and generated from the recommended studies should be used in a modeling approach to evaluate possible effects of PAHs on marine organisms. This model would be qualitatively similar to those recommended for trace metals and halogenated hydrocarbons.

5.6 UPTAKE, STORAGE, AND DEPURATION OF HALOGENATED HYDROCARBONS

This cateogry of studies has been assigned a high priority status for sewage sludge and dredge materials (which contain variable amounts of these compounds) and low priority status for other wastes. High priority status was assigned because: 1) these wastes can be significant sources of halogenated hydrocarbons; 2) information on the long-term fate and effects of these compounds is needed; and 3) these compounds are persistent and not rapidly metabolized (as are PAHs); consequently there is a greater likelihood for increased accumulation in tissues, transfer up the food chain, and effects on man. Recommended studies are presented below.

1) As with PAHs, the availability of halogenated hydrocarbons associated with marine sediments will depend on adsorption-desorption rates. As these hydrocarbons are desorbed into the interstitial pore water or near the surface of the sediments, they become more available to marine organisms. Additional information is needed on the rates of these processes. Experiments designed to examine these processes should be carried out in laboratory systems. The use of radiolabeled compounds would enhance the sensitivity of these experiments and would permit experiments in which organisms are exposed to a variety of compounds (each differentially labeled). Laboratory exposures should be comparable to probable ambient levels.

2) Some marine organisms have the ability to metabolize certain halogenated hydrocarbons (e.g., DDT, PCBs). This is important inasmuch as the long term effects of such compounds is related, in part, to their persistence. The limited information presently available indicates that the ability of marine organisms to metabolize, eliminate, or otherwise "detoxify" halogenated hydrocarbons is species dependent and compound dependent. This data base needs to be further developed in order to allow for a more complete evaluation of organisms' abilities to deal with accumulated halogenated hydrocarbons. We suggest that this information should be brought together in a logical fashion. This could involve developing information on major classes of halogenated hydrocarbons for major taxa (of fish and invertebrates).

Two basic kinds of studies are recommended. The first involves laboratory studies designed to measure rates of metabolism and composition of metabolites. The second includes careful tissue analyses of organisms taken from areas which have been exposed to contaminants and from noncontaminated areas. Organisms should be analyzed for the concentrations of compounds and metabolites. Collections should be made at major disposal areas.

3) There is evidence that halogenated hydrocarbons can induce MFO systems of organisms to metabolize PAHs. This could result in an increased rate of production of reactive metabolites. Assuming there is an adequate supply of PAHs to the organism, this effect could be important. At present, there is inadequate information with which to evaluate its significance. We recommend that laboratory studies be conducted to provide this information.

4) As indicated above, the effects of halogenated hydrocarbons will depend on a number of processes and the rates at which these occur. We recommend that a modeling approach be developed which incorporates these processes. This will assist in evaluating effects of these hydrocarbons discharged with ocean-dumped wastes.

5.7 UPTAKE, STORAGE, AND TRANSPORT OF RADIONUCLIDES

This cateogry of studies has been assigned high priority status for LLW and is not considered applicable to other waste categories. Based on our review of existing information the following studies are recommended.

1. Among the key concerns with regard to deep ocean disposal of LLW are possible biological and physical mechanisms which would transport radionuclides to the more productive surface layers. We recommend that existing information on these processes be reviewed and a status report prepared. The report should identify gaps in the information and recommend the studies on how the information should be obtained.

2. Existing predictive models should be evaluated for their adequacy in assessing public health and environmental risks associated with ocean disposal of LLW off the coasts of the United States. Two models which should be considered are the IAEA model used to assess continued suitability of LLW disposal in the northeast Atlantic and the model being developed as part of the U.S. Seabed Disposal Program. The IAEA model, which has been used to assess LLW disposal, is a generic rather than site-specific model. The Nuclear Energy Agency (NEA), which used the IAEA model, recognized that a site-specific model was the ideal but that at the time information was inadequate for developing such a model.

3. Site-selection criteria for disposal of LLW have been developed for the Pacific and Atlantic Oceans. These criteria will be used to select one or more candidate disposal areas. At present, there is little data on the environmental conditions of deep ocean (~4000 m) regions on which to completely evaluate the suitability of these areas as disposal sites. We recommend that once candidate sites are identified, appropriate field programs be implemented to provide necessary baseline information. These programs should focus on physical oceanographic, geological, chemical, and biological conditions at and above the benthic environment. Additional information should be obtained on physical and biological features of the water column inasmuch as these could be important in the

transport of radionuclides from the deep ocean to the more productive surface layers.

4. An assessment should be made of the integrity and anticipated lifetime of containers used to dispose of LLW. (IAEA has specified container design.) The consequences of container failure (in terms of potential radionuclide leakage) should be determined for various intervals (immediately, 1, 10, 100, 300, 1000 years). This information could be used to refine estimates of risks associated with LLW disposal at sea.

5.8 EFFECTS OF SEDIMENT ALTERATIONS ON BENTHIC LARVAL SETTLEMENT

This cateogry was assigned a high priority status for several of the waste types with significant solids content because: 1) there is evidence (e.g., studies with fly ash) that substrate modification can render sediments unattractive to settling benthic larvae, and; 2) there is little information on these kinds of effects. As discussed in subsection 4.2 of this report, settling benthic larvae respond to physical/chemical factors and, as a result, may avoid some substrates and select others. Substrates which have been modified as a result of dumping could experience diminished larval recruitment and consequently a reduction in benthic populations. These potential effects are not evaluated by current standard bioassay programs. We recommend that well-designed laboratory experiments be carried out to examine the effects of substrate modification by various waste types (e.g., fly ash, dredge materials, sludges) on benthic larval recruitment.

5.9 TOXICITY STUDIES (LETHAL/SUBLETHAL)

A number of the studies recommended in previous subsections involve laboratory studies designed to provide information on lethal and sublethal effects. Therefore, recommendations presented here are restricted to a few topics. Standard ocean dumping bioassay programs are designed to provide information on the relative acute toxicities of various wastes. Generally, these tests do not provide direct information on sublethal or chronic effects. However, these latter effects are of primary concern with regard to the continued use of the ocean as a waste disposal medium.

There have been [illegible] number of workshops (e.g., the recently held "Meaningful Measures in Marine Pollution") and there are a number of laboratories involved in addressing the problem of how to use laboratory microcosm/mecocosm systems to evaluate long-term and "ecosystem level" effects of ocean pollution. We believe that mesocosms, in particular, can provide a bridge between strictly laboratory studies and data generated from field programs. We recommend that studies using these systems continue in order to develop a base of information that can be used to relate results of field monitoring programs with those of more practical (less expensive, more rapidly performed, and better controlled) laboratory studies. Particular studies in which mesocosms could be used have already been discussed.

5.10 SITE MONITORING

Certain aspects of site monitoring for physical, chemical and biological conditions have already been presented in earlier sections. Those recommendations will not be repeated here. Site monitoring has been assigned a high priority status because field verification of the effects of ocean dumping is essential for sound waste management.

We recommend that the following aspects of site monitoring be considered.

1. Site monitoring programs should be designed to provide useful information. In most cases, a sampling design should be used that will provide data amenable to statistical analyses. A number of site monitoring and marine pollution monitoring programs which we have examined, have been ineffective in sampling appropriate parameters, have sampled parameters that are not useful or provide little information, have utilized inappropriate techniques, or have not allocated sampling effort in an efficient or useful manner. We recommend that future site monitoring programs be carefully designed with the assistance of a statistician with particular expertise in the allocation of sampling effort and with biological, chemical, and physical oceanographers familiar with current state-of-the-art techniques. Available information on existing sites provided by the U.S. EPA's past monitoring programs and information on the fate and effects of various contaminants should be used as a basis for determining what parameters should be measured in the future. Exhaustive "shotgun" sampling is expensive, time consuming, and can impair the implementation of well designed and focused programs.

2. To some extent (that is impossible to quantify) our present ability to monitor the marine environment, and, thus, assess the effects of waste disposal is limited by present sampling technology. For example, in the field of benthic ecology, we are using the same point sampling grab techniques that were first introduced over 50 years ago. In this field, as in other fields of biological and chemical oceanographic studies, there is a need for systems and instruments which can provide large scale reconnaissance as well as fine scale detail on environmental conditions in an efficient and cost-effective manner. Among the systems used more frequently for monitoring are various camera systems (e.g., sediment profile systems and water column shadow photography), high resolution side-scan, bat-fish water column profilers, and satellite remote sensing systems. We recommend that an assessment be made of recent developments in sampling technology in terms of how these could aid (perhaps more cost effectively) in the acquisition of site monitoring data. Of particular interest are remote sensing systems and systems which monitor benthic conditions.

5.11 INVENTORY OF WASTE CONSTITUENTS

This category of studies has been assigned high priority status for sewage sludge, dredged material, and industrial sludges because of the variability of these wastes and because studies have revealed (in particular for sewage sludge and dredged material) the presence of various chlorinated hydrocarbons which are of concern with regard to effects on marine organisms and public health. The constituents of fly ash, FGD sludge, and LLW are generally well known and waste constituent studies on these wastes have been assigned low priority status.

We recommend that available information on waste constituents be brought together. A number of data sources were identified as part of this program. A number of these have not been published. In particular, data on waste constituents in sewage sludge are available from the Association of Metropolitan Sewage Agencies (AMSA) and the Southern California Water Research Project (SCWRP). A number of chemical analytical studies have been initiated on waste constituents. Some (e.g., MacLeod et al., 1981), serve as reconnaissance surveys, providing preliminary information on constituents.

We recommend that state-of-the-art analytical techniques be used to identify and characterize the variability of constituents in wastes which may be ocean dumped. Of particular interest are measurements of halogenated hydrocarbons inasmuch as the composition, content, and variability of these are least known although they are of key concern with regard to potential effects.

5.12 CROSS-MEDIA RISK ASSESSMENT

This category of studies has been assigned high priority status because, in our opinion, it is essential to a technically sound waste management strategy. Environmental and public health risks of various alternatives for waste disposal were reviewed in Section 3 of this report. What is needed is the continued development of approaches (methodologies) for comparing the various alternatives in terms of their public health, environmental, social and economic implications.

Several cross-media risk analyses are presently underway as described in Section 3 of this report. These include EPA-funded programs involving sewage sludge and hazardous wastes, NRC-funded programs involving LLW, and several municipal-funded programs on sewage sludge. Our review of these programs indicated that most were in progress (i.e., the first few phases had been completed). We recommend that these programs, in particular those being funded by EPA, be carried through to completion. Further, we recommend that a mechanism be established to facilitate exchange of information among the various federally and state-funded programs. One possible way this might be accomplished is through a series of workshops. Our communications with the various principal investigators indicated that there had been little exchange of information. The products of these workshops would be a recommended approach, trial applications of this approach, a compilation of data required to implement the approach, and identification of gaps in information.

6. REFERENCES

Achinger, W.C., and L.E. Daniels. 1970. SW-51-ts-1j. U.S. Environmental Protection Agency.

Addison, R.R., M.F. Zinck, and D.E. Willis. 1977. Mixed function oxidase enzymes in trout (Salvelinus fontinalis) liver: absence of induction following feeding of p,p'-DDT or p,p'-DDE. Biochem. Physiol. 57C: 39-43.

Akin, E.W., W. Jakubowski, J.B. Lucas, and H.R. Pahren. 1978. Proc. Conference on Risk Assessment and Health Effects of Land Application of Municipal Wastewater and Sludges. University of Texas at San Antonio, Texas.

Akin, E.W., W.H. Benton, and W.F. Hill, Jr. 1971. Thirteenth Water Quality Conference, University of Illinois: 59.

Amstutz, R.W. 1980. Deep-well disposal: A valuable natural resource. In: Toxic and Hazardous Waste Disposal. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan 4: 313.

Anderson, A.R., P.W. Anderson, W.M. Dunstan, L.L. Falk, J.E. Kerrigan, C.D. Rose, and R. Tollefson. 1979. Sources. In: Proceedings of a Workshop on the Assimilative Capacity of U.S. Coastal Waters for Pollutants. Crystal Mountain, Washington. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Anderson, D.C., D.W. Brown, and J. Green. 1981. Organic leachate effects on the permeability of clay liners. Presented at the National Conference on Management of Uncontrolled Hazardous Waste Sites, Washington, D.C. October 28-30, 1981.

Antsyferov, S.M., and R.D. Kos'yan. 1980. Sediments suspended in stream flow. ASCE Proc. JHD, 106 (HY2): 313-330.

Ayers, R.C., Jr., T.C. Sauer, Jr., D.O. Stuebner, and R.P. Meek. 1980. An environmental study to assess the effect of drilling fluids on water quality parameters during high rate, high volume discharges to the ocean. In: Proc. Symposium Research on Environmental Fate and Effects of Drilling Fluids and Cuttings. Lake Buena Vista, Florida. January 21-24, 1980.

Bayne, B.L., D.A. Brown, F.L. Harrison, and P.P. Yevich. 1980. Mussel health. In: International Mussel Watch, Report on a Workshop at Barcelona, Spain, December 1978. National Academy of Science, Washington, D.C.

Beard, P.J. 1938. J. Am. Wat. Wks Ass. 30: 124.

Becker, W.F., A.A. Moghissi, F.H.F. Au, E.W. Brethauer, and J.C. McFarlane. 1974. Nature (London) 249: 674.

Billen, G. 1978. A budget of nitrogen recycling in North Sea sediments off the Belgian coast. Est. Coastal Mar. Sci. 7: 127-146.

Boehm, P.D., and J.S. Quinn. 1973. Solubilization of hydrocarbons by the dissolved organic matter in sea water. Geochimica and Cosmochimica Acta 37: 2459-2477.

Booz, Allen, and Hamilton. 1982. Multi-media risk assessment of municipal sludge disposal options; Task 1 - Report on data availability. Booz, Allen, and Hamilton, Inc.

Boswell, F.C. 1975. J. environ. Qual. 4: 267.

Brandsma, M.G., and D.J. Divoky. 1976. Development of models for prediction of short-term fate of dredged material discharged in the estuarine environment. Contract Report D-76-5, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Brady, F.O., M. Panemangalore, F.A. Day, A.M. Fiskin, and G. Peterson. 1979. In vivo and ex vivo induction of rat liver metallothionein. In: Metallothionein: Proceedings of the First International Meeting on Metallothionein and other Low Molecular Weight Metal-Binding Proteins, J.H.R. Kaji and M. Nordberg (eds.). Brikhauser Verlag, Boston, Massachusetts.

Bremner, I., and N.T. Davies. 1975. The induction of metallothionein in rat liver by injection and restriction of food intake. Biochem. J. 149: 733-738.

Brown, D.A., R.W. Gossett, and K.D. Jenkins. 1981. Contaminants in white croakers (Genyonemus lineatus) from the southern California Bight: II. Xenobiotic hydrocarbon detoxification. Submitted for publication in Fifth Symposium on Pollution and Physiology of Marine Organisms. Academic Press, New York.

Brown, K.F. 1981. Crab meal production: Tragic impact on the blue crab industry unless viable alternatives are established. In: W.S. Otwell (ed.): 280-284.

Bryan, G.W. 1980. Recent trends in research on heavy metal contamination in the sea. Helogoländer Meeresunters 33: 6-25.

Bryan, G.W. 1979. Bioaccumulation of marine pollutants. Phil. Trans. R. Soc. Lond B. 286: 483-505.

Califano, R.J. 1981. Accumulation and tissue distribution of polychlorinated biphenyls (PCBs) in early life stages of the striped bass, Morone saxatilis. Ph.D. Dissertation. New York University, New York. 163 pp.

CEC. 1979. Power-reactor low-level radioactive waste generation and waste management practices evaluation; Report 2 - Policy recommendations, conclusions, and summary. Prepared by NUS Corporation, Palo Alto, California, for the California Energy Commission.

CEQ. 1981a. The global 2000 report to the President. Volume two. The technical report. The Council on Environmental Quality and the Department of State.

CEQ. 1981b. The global 2000 report to the President. Volume three. Documentation on the government's global sectoral models: The government's "global model." The Council on Environmental Quality and the Department of State.

CEQ. 1981c. Environmental quality - 1980. Council on Environmental Quality, Washington, D.C.

Champ, M.A., and P.K. Park. 1981. Ocean dumping of sewage sludge: A global review. Sea Technology 22: 18-24.

Chaney, R.L. 1974. Agr. Res. Serv., U.S. Department of Agriculture, Beltsville, Maryland.

Chass, R.L., and A.H. Rose. 1953. Air Repair 3: 119.

Chen, K.Y., and F.R. Bowerman. 1974. In: T.F. Yen (ed.), Recycling and disposal of solid wastes. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan: 349-367.

Chia, F.S., and M.E. Rice (eds.). 1977. Settlement and metamorphosis of marine invertebrate larvae. Elsevier, New York. 290 pp.

Christensen, R.L. 1977. Massachusetts Agriculture Experimental Station, College of Food and Natural Resources, University of Massachusetts, Amherst, Massachusetts.

Cole, H.A. 1979. Pollution of the sea and its effects. Proc. R. Soc. Lond B. 205: 17-30.

Council for Agricultural Science and Technology. 1981. EPA 600/8-81-003. Ames, Iowa.

Craig, P.J. 1980. Metal cycles and biological methylation. In: The Handbook of Environmental Chemistry, Vol. 1A. Springer-Verlag, Berlin.

Cram, E.B. 1943. Sew. Works J. 15: 1119.

Curl, H., R. Dugdale, E. Jenne, D.P. Larsen, R. Lasker, J.M. De Manche, C. Officer, J. Ryther. 1979. Biostimulants in scientific problems relating to ocean pollution. E.D. Goldberg (ed.). NOAA.

Dacre, J.C. 1980. Potential health hazards of toxic organic residues in sludge. In: Sludge Risks of Land Application. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan. 367 pp.

Dayal, R., A. Okubo, I.W. Duedall, and A. Ramamoorthy. 1979. Radionuclide redistribution mechanisms at the 2800-m Atlantic nuclear waste disposal site. Deep-Sea Res. 26A: 1329-1345.

Dean, R.B. 1973. Proc. Conference on Land Disposal of Municipal Effluents and Sludges. Rutgers University, New Jersey.

DOE. 1981a. Cost and quality of fuels for electric utility plants - July 1981. U.S. Department of Energy, Washington, D.C.

DOE. 1981b. Cost and quality of fuels for electric utility plants - 1980 Annual. U.S. Department of Energy, Washington, D.C.

DOE. 1980. Cost and quality of fuels for electric utility plants - 1979 Annual. U.S. Department of Energy, Washington, D.C.

DOE. 1979. Cost and quality of fuels for electric utility plants - 1978 Annual. U.S. Department of Energy, Washington, D.C.

Doran, J.W. 1976. Ph.D. Dissertation, Cornell University, Ithaca, New York.

Doran, J.W., and M. Alexander. 1975. Ann. Mtg. Amer. Soc. Microbiol. New York.

Doran, J.W., J.R. Ellis, and T.M. McCalla. 1977. In: Land as a Waste Management Alternative, R.C. Loehr (ed.). Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan: 343-362.

Duprey, R.L. 1968. Public Health Service Publication No. 999 AP-42.

Dyer, R.S. 1980. Sea disposal of nuclear waste: A brief history. In: Nuclear Waste Management, T.C. Jackson (ed.). Pergamon Press, New York. 124 pp.

Eberhardt, H., and W. Mayer. 1968. Proc. National Incinerator Conf., New York, Amer. Soc. Mechanical Engr.: 76-77.

Eliasson, R. 1972. Eng. News Record 128: 912.

Engle, D.W., and B.A. Fowler. 1979. Factors influencing cadmium accumulation and its toxicity to marine organisms. Health Perspectives 28: 81-88.

EPA. 1979. Economic impact analysis of hazardous waste management regulations on selected industries. Prepared by Energy Resources Company, Cambridge, Massachusetts, for the U.S. Environmental Protection Agency. December.

EPA. 1977a. Industrial process profiles for environmental use: Chapter 22. The phosphate rock and basic fertilizer materials industry. U.S. Environmental Protection Agency, Cincinnati, Ohio.

EPA. 1977b. Industrial process profiles for environmental use: Chapter 26. Titanium industry. U.S. Environmental Protection Agency, Cincinnati, Ohio.

EPA. 1974. Polluted groundwater: Estimating the effects of man's activities. U.S. Environmental Protection Agency, Las Vegas, Nevada.

EPRI. 1979. FGD sludge disposal manual. Prepared by Michael J. Baker, Jr., Inc., Beaver, Pennsylvania.

Epstein, E. 1975. J. environ. Qual. 4: 139.

Epstein, E., J.M. Taylor, and R.L. Chaney. 1976. J. environ. Qual. 5: 422.

Ewald, W.G., J.E. French, and M.A. Champ. 1976. Toxicity of polychlorinated biphenyls to Euglena gracilis cell population growth, carbon fixation, chlorophyl level, oxygen consumption, and protein and nucleic acid synthesis. Bull. environ. Contamin. Toxicol. 16: 71-80.

Farrell, J.B. 1974. Overview of sludge handling and disposal. In: Proceedings of the National Conference on Municipal Sludge Management, Pittsburgh, Pennsylvania.

Farrington, J.W. 1981. Biogeochemistry of PCBs in New Bedford Harbor and Buzzards Bay, Massachusetts. Woods Hole Oceanographic Institution, Chemical Processes and Pollution. Sea Grant Proposal.

Feldt, W., G. Kanisch, and R. Lauer. 1980. Radioactive contamination of NEA dumping sites. IAEA Symposium on the Impacts of Radionuclide Release into the Marine Environment. Draft report. U.S. Environmental Protection Agency.

Fingerman, S.W., and M. Fingerman. 1979. Comparison of the effects of fourteen-day and chronic exposures to a polychlorinated biphenyl, Arochlor 1242, on molting of the fiddler crab, Uca pugilater. Bull. environ. Contamin. Toxicol. 21: 352-357.

Fisher, N.S. 1975. Chlorinated hydrocarbon pollutants and photosynthesis of marine phytoplankton: A reassessment. Science 189: 463.

Fogg, G.E. 1982. Nitrogen cycling in sea waters. Phil. Trans. R. Soc. Lond. B29654-520.

Förstner, U., and G.T.W. Wittman. 1981. Metal pollution in the aquatic environment. Springer-Verlag. Berlin, Heidelberg, New York. 486 pp.

Frank, R., M. Holdrinet, E.H. Braun, R.L. Thomas, A.W.L. Kemp, and J.J. Jaquet. 1977. Organochlorines, insecticides and PCBs in sediments of Lake St. Clair (1970 and 1974) and Lake Erie (1971). Science of the total environment, Ontario Ministry of Agriculture and Food, Canada 8(3): 205.

GAO. 1981. Billions could be saved through waivers for coastal wastewater treatment plants CED-81-68. U.S. General Accounting Office, Gaithersberg, Maryland.

GAO. 1978. Sewage sludge - How do we cope with it? U.S. General Accounting Office report to Congress, CED-78-152.

Goldberg, E.D. (ed.). 1980. Proceedings of a Workshop on Assimilative Capacity of U.S. Coastal Waters for Pollutants. Crystal Mountain, Washington, July 29 - August 4, 1979. Second Printing, Revised. U.S. Department of Commerce, Boulder, Colorado.

Green, W.J., G.F. Lee, and R.A. Jones. 1981. Clay-soils permeability and hazardous waste storage. JWPCF 53(8).

Greenberg, A.E., and E. Kupka. 1957. Sew. and Indust. Wastes, 29: 524. Gruger, E.H., Jr., T. Hruby, and N.L. Karrick. 1976. Sublethal effects of structurally related tetrachloro-, pentachloro-, and hexachlorobiphenyl on juvenile coho salmon. Environ. Sci. Technol. 10: 1033-1037.

Haedrich, R.L., and N.R. Henderson. 1974. Pelagic food of Coryphaenoids armatus, a deep benthic rattail. Deep-Sea Res. 21: 739-744.

Halebsky, M. 1980. Ocean incineration of toxic chemical wastes. In: Toxic and Hazardous Waste Disposal. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan 4: 313.

Halter, M.T., and H.E. Johnson. 1977. A model system to study the description and biological availability of PCBs in hydrosails. Presented at ASTM Aquatic Toxicology 1st Annual Sym., Memphis, P(178) (18).

Hansen, W.G., and H.L. Rishel. 1981. Cost comparisons of treatment and disposal alternatives for hazardous materials, project summary. EPA-600/52-80-188. U.S. Environmental Protection Agency, Cincinnati, Ohio.

Hinz, R., and F. Matsumura. 1977. Comparative metabolism of PCB isomers by three species of fish and the rat. Bull. environ. Contamin. Toxicol. 18: 631-639.

Hutchinson, T.C., J.A. Hellebust, D. MacKay, D. Tam, and P. Kauss. 1979. Relationship of hydrocarbon solubility to toxicity in algae and cellular membrane effects. In: API, EPA, and USCG, 1979 Oil Spill Conf. (Prevention, Behavior, Control, Cleanup). American Petroleum Institute, Washington, D.C.

Hyde, H. 1981. Incineration (thermal reduction). In: The Sludge Management Syndrome: Burn, Barge, Bury, or Bust! Proceedings of the Association of Metropolitan Sewage Agencies, Eastsound, Washington.

IAEA. 1978a. The oceanographic basis of the IAEA revised definition and recommendations concerning high level radioactive waste unsuitable for dumping at sea. IAEA-210, International Atomic Energy Agency, Vienna, Austria. 59 pp.

IAEA. 1978b. The radiological basis of the IAEA revised definition and recommendations concerning high-level radioactive waste unsuitable for dumping at sea. IAEA-211. International Atomic Energy Agency, Vienna, Austria. 58 pp.

Interagency Committee on Ocean Pollution, Research, Development and Monitoring. 1981. National Marine Pollution Program Plan. Federal Plan for Ocean Pollution Research, Development, and Monitoring. Fiscal Years 1981-1985. NOAA, Environmental Research Laboratories, Boulder, Colorado.

Iseki, K., M. Takahashi, E. Bauerfeind, and C.S. Wong. 1981. Effects of polychlorinated biphenyls on a marine phytoplankton population and sedimentation in controlled ecosystem enclosures. Marine Ecology - Progress Series 5: 207-214.

Jenkins, K.D., D.A. Brown, G.P. Hershelman, and C. Meyer. 1981. Contaminants in white croakers (Genyonemus lineatus) from the southern California Bight: I. Trace metal detoxification. Submitted for publication in Fifth Symposium on Pollution and Physiology of Marine Organisms. Academic Press, New York.

Jens, W., and F.R. Rehm. 1966. Proc. National Incinerator Conf., New York, Amer. Soc. Mechanical Engr.: 74-83.

Johnson, R.D., R.L. Jones, T.D. Hinesly, and D.J. David. 1974. Selected chemical characteristics of soil, forages, and drainage water from the sewage farm serving Melbourne, Australia. Department of the Army Corps of Engineers.

Josephson, J. 1976. Environ. Sci. Technol. 10: 226.

Kaji, J.H.R., and M. Nordberg. 1979. Metallothionein: Proceedings on the First International Meetings on Metallothionein and other Low Molecular Weight Metal-Binding Proteins, J.H.R. Kaji and M. Norberg (eds.). Brikhauser Verlag, Boston, Massachusetts.

Keeney, D.R., and R.E. Wildung. 1977. In: Soils for Management of Organic Wastes and Waste Waters. ASA, CSSA, SSSA, Madison, Wisconsin: 75-97.

Ketchum, B.H., D.R. Kester, and P.K. Park (eds.). 1981. Ocean Dumping of industrial wastes. Proceedings of the First International Ocean Dumping Symposium, University of Rhode Island, West Greenwich, Rhode Island, 10-13 October 1978. Plenum Press, New York. 525 pp.

Klein. 1974.

Koh, R.C.Y., and Y.C. Chang. 1973. Mathematical model for barged ocean disposal of wastes. EPA-660/2-73-029. Prepared for Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C.

Krone, R.B. 1968. Proc. Symposium on the Use of Municipal Sewage Effluent for Irrigation, Louisiana Polytechnic Institute, Ruston, Louisiana.

Krone, R.B., G.T. Orlob, and C. Hodkinson, 1958. Sew. and Indust. Wastes 30: 1.

Lee, G.F. 1977. Contamination in the New York Bight. Discussion of the paper by J.A. Mueller, A.R. Anderson, J.S. Jeris (1976). J. Wat. Pollut. Control Fed. 49: 1920-1921.

Lee, G.F., R.A. Jones, F.Y. Saleh, G.M. Mariani, D.H. Hanover, J.S. Butler, and P. Bandyopadhzy. 1978. Evaluation of the elutriate test as a method of predicting contaminant release during open water disposal of dredged sediment and environmental impact of open water dredged material disposal. Final Report, prepared for Office, Chief of Engineers, U.S. Army, Washington, D.C. 1186 pp.

Lee, G.F., J.M. Lopez, and G.M. Mariani. 1975b. Leaching and bioassay studies on the significance of heavy metals in dredged sediments. Presented at the International Conference on Heavy Metals in the Environment: 731-764.

Lee, G.F., and G.M. Mariani. 1977. Evaluation of the significance of waterway sediment associated contaminants on water quality at the dredged material disposal site. Aquatic Toxicology and Hazard Evaluation, ASTM STP 634: 196-213.

Letey, J., Jr. 1977. In: Soils for Management of Organic Wastes and Waste Waters. ASA, CSSA, SSSA, Madison, Wisconsin: 101-112.

Lieber, M., and W.F. Welsch. 1954. J. Am. Wat. Wks Ass. 46: 541.

Lipsky, M.M., J.E. Klannig, and D.E. Hinton. 1978. Comparison of acute response to polychlorinated biphenyl in liver of rat and channel catfish: A biochemical and morphological study. J. Toxicol. environ. Hlth 4: 107-121.

LLWMP. 1980a. Managing low-level radioactive waste, a proposed approach LLWMP-1. Low-level Waste Management Program, EG&G Idaho, Inc., Idaho Falls, Idaho. 90 pp.

LLWMP. 1980b. Understanding low-level radioactive waste, LLWMP-2. Low-level Waste Management Program, EG&G Idaho, Inc., Idaho Falls, Idaho.

Lopez, J.M. 1977. Significance of heavy metals associated with dredged materials. Ph.D. Dissertation, University of Texas at Dallas, 220 pp.

Lund, L.J., A.L. Page, and C.O. Nelson. 1974. Agron. Abstr. 66: 33.

MacBeth, P., G. Wehmann, B. Thamer, and D. Card. 1979. Evaluation of alternative methods for the disposal of low-level radioactive waste. U.S. Nuclear Regulatory Commission. NRC FIN No. B1521.

MacCave, I.N. 1978. Sediments in the abyssal boundary layer. Oceanus 21: 29-33.

MacLean, A.J., B. Stone, and W.E. Cordukes. 1973. Can. J. Soil Sci. 53: 130.

MacLeod, W.D., Jr., L.S. Ramos, A.J. Friedman, D.G. Burrows, P.G. Prohaska, D.L. Fisher, and D.W. Brown. 1981. Analysis of residual chlorinated hydrocarbons, aromatic hydrocarbons, and related compounds in selected sources, sinks, and biota of the New York Bight. NOAA Technical Memorandum OMPA-6. National Oceanographic and Atmospheric Administration (NOAA), Boulder, Colorado. 128 pp.

Mallman, W.L., and W. Litsky. 1951. Amer. J. publ. Hlth 41: 38.

Maurer. J., J. Calambokides, N. Musgrove, B. Drager, M. Berg, and S.G. Herman. 1977. Polychlorinated biphenyls in cottids, mussels, and sediment in southern Puget Sound, Washington. Bul. environ. Contamin. Toxicol. 18: 588.

Maxwell, M.A., et al. 1978. Sulfur oxides control technology in Japan. Prepared for Henry M. Jackson, Chairman, Senate Committee on Energy and Natural Resources.

May, T.G., and T.J. McManus. 1977. Environmental and economic assessment of effluent standards for a chemical facility. Plymouth Meeting, Pennsylvania.

McCain, B.B., H.O. Hodgins, W.D. Gronlund, J.W. Hawkes, D.W. Brown, M.S. Myers, and J.H. Vandermeulen. 1978. Bioavailability of crude oil from experimentally oiled sediments to English sole (*Parophrys vetulus*), and pathological consequences. J. Fish. Res. Bd Can. 35: 657.

McCalla, T.M., J.R. Peterson, and C. Lue-Hing. 1977. *In*: Soils for Management of Organic Wastes and Waste Waters. ASA, CSSA, SSSA, Madison, Wisconsin: 11-41.

McDermott, D.J., D.R. Young, and T.C. Heesen. 1975. Polychlorinated biphenyls in marine organisms off southern California. Southern California Coastal Water Research Project, Summary Report, 1971-1975, El Segundo, California, 49 pp.

McLeese, D.W., C.D. Metcalfe, and D.S. Pezzack. 1980. Bioaccumulation of chlorobiphenyls and endrin from food by lobsters (*Homarus americanus*). Bull. environ. Contamin. Toxicol. 25: 161-168.

Menzie, C.A., D. Maurer, and W.A. Leathem. 1980. An environmental monitoring study to assess the impact of drilling discharges in the Mid-Atlantic. IV. The effects of drilling discharges on the benthic community. *In*: Proc. Symp. Res. Environ. Fate and Effects of Drilling Fluids and Cuttings, Lake Buena Vista, Florida, January 21-24, 1980.

Merz, R.C., and R. Stone. 1969. Special studies of sanitary landfills, final summary report (January 1, 1964 - December 31, 1968) to USPHS, HEW. University of Southern California, Los Angeles, California.

Moore, S.A., and R.C. Harris. 1972. Effects of polychlorinated biphenyls (A 1242) on marine phytoplankton communities. Nature 240: 356-358.

Morel, F.M., and S.L. Schiff. 1980. Geochemistry of municipal waste in coastal waters. Massachusetts Institute of Technology, Dept. of Civil Engineering Report No. 259.

Mosier, A.R., S.M. Morrison, and G.K. Elmund. 1977. *In*: Soils for Management of Organic Wastes and Waste Waters. ASA, CSSA, SSSA, Madison, Wisconsin: 531-571.

Mueller, J.A., J.S. Jeris, A.R. Anderson, and C.F. Hughes. 1976. Contaminant inputs to the New York Bight. NOAA TM ERL MESA-6. Marine Ecosystems Analysis Program Office, Boulder, Colorado, 347 pp.

Murakami, K., and K. Takeishi. 1976. Behavior of heavy metals and PCBs in the removal and treatment operations of bottom deposits. Presented at Toxic Substances 2nd Experts Meeting, Tokyo, Japan. Ministry of Construction.

Murphy, L.S., P.R. Hoar, and R.A. Belastock. 1981. The effects of industrial wastes on marine phytoplankton. *In*: Ocean Dumping of Industrial Wastes, B.H. Ketchum, D.R. Kester, and P.K. Park (eds.), Plenum Press, New York and London. 525 pp.

Murray, T.J. 1981. Seafood waste management in Virginia. In: W.S. Otwell (ed.): 3-9.

Musick, J.A., and S.J. Sulak. 1978. Characterization of the demersal fish community of a deep sea radioactive dump site. Draft. Submitted to the U.S. Environmental Protection Agency.

NACOA. 1981. The role of the ocean in a waste management strategy. A special report to the President and the Congress. National Advisory Committee on Oceans and Atmosphere, Washington, D.C. 103 pp.

National Ash Association. 1982. Personal communication.

National Research Council. 1976. Disposal in the marine environment - an oceanographic assessment. 76 pp.

Nebert, D.W., S.W. Bigelow, A.B. Okey, T. Yahagi, Y. Mori, M. Nagao, and Y. Sugimura. 1979. Pyrolysis products from amino acids and protein: highest mutagenicity requires cytochrome P,-450. Proc. natn. Acad. Sci. U.S.A. 76: 5929.

Neff, J.M. 1979. Polycyclic aromatic hydrocarbons in the aquatic environment. Applied Science Publishers, London. 262 pp.

Niessen, W.R. 1970. NTIS Report PB 192-378. Springfield, Virginia.

Nixon, S.W. 1981. Remineralization and nutrient cycling in coastal marine ecosystems. In: Estuaries and Nutrients, B.J. Nielson and E.L. Cronin (eds.). Humana Press, Inc. Clifton, N.J.

NMFS. 1973-1981. Fisheries statistics of the United States. National Marine Fisheries Service, Washington, D.C.

NOAA. 1979a. Assimilative capacity of U.S. coastal waters for pollutants. E.D. Goldberg (ed.), U.S. Department of Commerce, NOAA Environmental Research Laboratories, Boulder, Colorado. 284 pp.

NOAA. 1979b. Chemical pollutants of the New York Bight: Priorities for research. J.S. O'Connor and H.M. Stanford (eds.). U.S. Department of Commerce, NOAA Environmental Research Laboratories, Boulder, Colorado. 217 pp.

NOAA. 1975. Ocean dumping in the New York Bight. NOAA Technical Report ERL 321-MESA-2. Boulder, Colorado. 78 pp.

NRC. 1977. Multimedium management of municipal sludge. National Research Council, Washington, D.C.

Nuclear Energy Agency. 1980. Review of the continued suitability of the dumping site for radioactive waste in the Northeast Atlantic. Nuclear Energy Agency, Organization for Economic Cooperation and Development. Paris, France. 100 pp.

NUS. 1980. The 1979 state-by-state assessment of low-level radioactive wastes shipped to commercial burial grounds. Prepared by NUS Corporation, San Francisco, California.

Okubo, A. 1971. Ocean diffusion diagrams. Deep-Sea Res. 18: 789-802.

Orr, M.H. 1981a. Qualitative dispersion characteristics of the particulate phase of Dupont Edgemoor Plant acid-iron waste and grassell: plant wastes for summer and winter. In: Assessment Report on the Effects of Waste Dumping in 106-Mile Ocean Waste Disposal Site: Dumpsite Evaluation Report 81-1. NOAA Special Report, Boulder, Colorado.

Orr, M.H. 1981b. Qualitative dispersion characteristics of sewage sludge released in the presence of a shallow seasonal thermocline. In: Assessment Report on the Effects of Waste Dumping in 106-Mile Ocean Waste Disposal Site: Dumpsite Evaluation Report 81-1. NOAA Special Report, Boulder, Colorado.

Osag, T.R., and G.B. Crane. 1974. EPA-450/1-74-006. U.S. Environmental Protection Agency.

OTA. 1979a. The direct use of coal - prospects and problems of production and combustion. Office of Technology Assessment, Congress of the United States, Washington, D.C.

OTA. 1979b. The direct use of coal. Office of Technology Assessment, Congress of the United States, Washington, D.C. 411 pp.

Otwell, W.S. (ed). 1981. Seafood waste management in the 1980s. Conference Proceedings, Orlando, Florida, September 23-25, 1980. Report Number 40. Florida Sea Grant College, Gainesville, Florida. 365 pp.

Page, A.L., and A.C. Chang. 1975. Proc. Second Nat. Conf. Water reuse: Water's interface with energy, air, and solids, Chicago, Illinois. Sponsored by Amer. Inst. Chem. Eng. and U.S. Environmental Protection Agency. Tech. Transfer.

Pederson, D.C. 1981. EPA-600/52-81-170. Contract No. 68-03-2803. U.S. Environmental Protection Agency.

Pequegnat, W.E., D.D. Smith, R.M. Darnell, B.J. Presley, and R.O. Reid. 1978. An assessment of the potential impact of dredged material disposal in the open ocean. Dredged Material Research Program Technical Report D-78-2. Office, Chief of Engineers, U.S. Army, Washington, D.C. 642 pp.

Perry, A. 1981. Seafood waste management in the Gulf of Mexico. In: W.S. Otwell (ed.): 72-77.

Peterson, R.E., and P.D. Guiney. 1979. The disposition of polychlorinated biphenyls in fish. In: Pesticide and Xenobiotic Metabolism in Aquatic Organisms, M.A.Q. Kahn, J.J. Lech, and J.J. Menn (eds.). American Chemical Society Symposium Series 99, Washington, D.C.

Pigott, G.M. 1981. Seafood waste management in the northwest and Alaska. In: W.S. Otwell (ed.): 78-95.

Pliss, G.B., and V.V. Khudoley. 1975. Tumor induction by carcinogenic agents in aquarium fish. J. Natl. Cancer Inst. 55: 129.

PneumoDynamics Corporation. 1961. Technical report: Survey of radioactive waste disposal sites. Prepared under Contract AT(04-3)-331 for the U.S. Atomic Energy Commission. July.

Proni, J.R., and D.V. Hansen. 1981. Dispersion of particulates in the ocean studied acoustically: The importance of gradient surfaces in the ocean. In: Ocean Dumping of Industrial Wastes. B.H. Ketchum, D.R. Kester, and P.K. Park (eds.), Plenum Press, New York. 525 pp.

Quraishi, M.S. 1977. Biochemical insect control. John Wiley and Sons, New York.

Rains, B.A., M.J. DePrimo, and I.L. Groseclose. 1973. EPA-670/2-73-098. Grant No. WPRD 23-01-68. U.S. Environmental Protection Agency.

Reed, A.W. 1975. Ocean waste disposal practices. Noyes Data Corporation, Park Ridge, New Jersey.

Reimers, R.S., M.D. Little, A.J. Englande, D.B. Leftwich, D.D. Bowman, and R.F. Wilkinson. 1981. EPA-600/52-81-166. Grant No. 805107. U.S. Environmental Protection Agency.

Reish, D.J. 1977. Survey of the benthic invertebrates collected from the United States radioactive waste disposal sites in the Atlantic and Pacific oceans. Draft. Prepared for the U.S. Environmental Protection Agency by Reish Marine Studies, Inc., Los Alamitos, California.

Richards, M.P., and R.J. Cousins. 1975. Influence of parenteral zinc and actinomycin D on tissue zinc uptake and the synthesis of a zinc binding protein. Bioinorg. Chem. 4: 215-224.

Romero, J.C. 1970. Ground Water 8: 37.

Ross, D.E., and H.T. Phung. 1980. Soil incorporation (landfarming) of industrial wastes. In: Toxic and Hazardous Waste Disposal. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan 4: 313.

Ryther, J.H., and W.M. Dunstan. 1971. Nitrogen, phosphorous and eutrophication in the coastal marine environment. Science, 141: 1008-1013.

Sanborn, J.R., W.F. Childers, and R.L. Metcalf. 1975. Uptake of three polychlorinated biphenyls, DDT and DDE by the green sunfish, Lepomis cyanellus rat. Bull. environ. Contamin. Toxicol. 13: 209-217.

Smayda, T.J. 1970. The suspension and sinking of phytoplankton in the sea. Oceanogr. mar. Biol. Rev. 8: 353-414.

Smith, C.E., T.H. Peck, R.J. Klauda, and J.B. McLaren. 1979. Hepatomas in Atlantic tomcod Microgadus tomcod (Walbaum) collected in the Hudson River estuary in New York. J. Fish. Dis. 2: 313.

Smith, J.D. 1977. Modeling of sediment transport on continental shelves. In: The Sea, Vol. 6. E.D. Goldberg, I.N. McCave, J.J. O'Brien, and J.H. Steele (eds.), Wiley-Interscience, New York.

Stabenow, G. 1972. Proc. National Incinerator Conf., New York. Amer. Soc. Mechanical Engr.: 178-194.

Stegeman, J.J. 1981. Polynuclear aromatic hydrocarbons and their metabolism in the marine environment. In: Polycyclic Hydrocarbons and Cancer, H.V. Gelboin and P.O.P. Ts'o (eds.). Academic Press, New York.

Stoffers, P., C. Summerhayes, U. Förstner, and S.R. Patchineelam. 1977. Copper and other heavy metal contamination in sediments from New Bedford Harbor. Woods Hole Oceanographic Institution, Woods Hole, Massachusetts.

Stone, R. 1977. In: Handbook of Solid Waste Management, D.G. Wilson (ed.). Van Nostrand Reinhold Company, New York, New York: 226-263.

Sullivan, R.J. 1969. Dept. of Health, Educ. and Welfare. No. PH-22-68-25.

Tagatz, M.E., J.M. Ivey, H.K. Lehman, M. Tobia, and J.L. Oglesby. 1980. Effects of a drilling fluid on development of experimental estuarine macrobenthic communities. In: Proc. Symp. Res. Environ. Fate and Effects of Drilling Fluids and Cuttings, Lake Buena Vista, Florida, January 21-24, 1980.

Tennessee Valley Authority. 1979. North American production capacity data. National Fertilizer Development Center.

Thomas, J.P., J.E. O'Reilly, A.F. Draxler, J.A. Babinchak, C.N. Robertson, W.C. Phoel, R.I. Waldhaver, C.A. Evans, A. Matte, M.S. Cohn, M.F. Nitkowski, and S. Dudley. 1979. Biological processes: Productivity and respiration. In: Oxygen Depletion and Associated Benthic Mortalities in New York Bight, 1976, R.L. Swanson and C.J. Sondermann (eds.). NOAA Professional Paper.

Thomas, W.H., and D.L.R. Seibert. 1977. Effects of copper on the dominance and the diversity of algae: Controlled ecosystem pollution experiment. Bull. mar. Sci. 27: 23-33.

Trees, W.E., G.L. Roberts, and C.R. Priesing. 1979. Neutralization of sulfate process titanium dioxide wastes with limestone and lime. J. Wat. Pollut. Control Fed. 51: 158-162.

U.S. Army Corps of Engineers. 1979. Disposal of dredged material within the New York District: Volume 1, Present practices and candidate alternatives. Prepared by the MITRE Corporation, McLean, Virginia.

U.S. Bureau of Mines. 1982. Personal communication.

U.S. Bureau of Mines. 1976. Cited in Coal reserves data reported in OTA, 1979.

U.S. Bureau of Mines. 1975. Cited in Coal reserves data reported in OTA, 1979.

U.S. Nuclear Regulatory Commission. 1979. The environmental effects of using coal for generating electricity. Draft. Argonne, Illinois.

Veith, E.D. 1972. Recent fluctuations of chlorobiphenyls (PCBs) in the Green Bay, Wisconsin region. Environ. Health Perspectives 1: 51-54.

Vogt, J.E. 1961. Proc. 1961 Symp. on Groundwater Contamination. USPHS Tech. Rept. W61-5: 87.

Walker, A.B. 1964. Proc. National Incinerator Conf., New York, Amer. Soc. Mechanical Engr.: 13-19.

Walker, A.B., and F.W. Schmitz. 1966. Proc. National Incinerator Conf., New York, Amer. Soc. Mechanical Engr.: 64-73.

Walsh, J.J., G.T. Rowe, R.L. Iverson, and C.P. McRoy. 1981. Biological export and shelf carbon is a sink of the global CO_2 cycle. Nature (London) 291: 196-201.

Webb, et al. 1976. A pilot research investigation and demonstration on the processing and marketing of selected fishery species in the coastal plains area. Coastal Plains Center for Marine Development Services. Cited in R.E. Caruvan and F.B. Thomas. 1981. In: W.S. Otwell (ed.): 10-25.

Weinstein, N.J., and R.F. Toro. 1976. Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan.

Wilson, A.J., and J. Forestor. 1978. Persistence of arochlor (trade name) 1254 in a contaminated estuary. Bull. environ. Contamin. Toxicol. 19: 637-640.

Wong, P.T.S., Y.K. Chan, and P.L. Luxon. 1975. Nature (London) 253: 263.

Wood, J.M. 1974. Biological cycles for toxic elements in the environment. Science 183: 1049-1052.

APPENDIX A

NATIONAL MARINE POLLUTION CONCERNS REGARDING MARINE WASTE DISPOSAL FROM THE NATIONAL MARINE POLLUTION PROGRAM PLAN: FEDERAL PLAN FOR OCEAN POLLUTION RESEARCH, DEVELOPMENT, AND MONITORING FISCAL YEARS 1981-1985

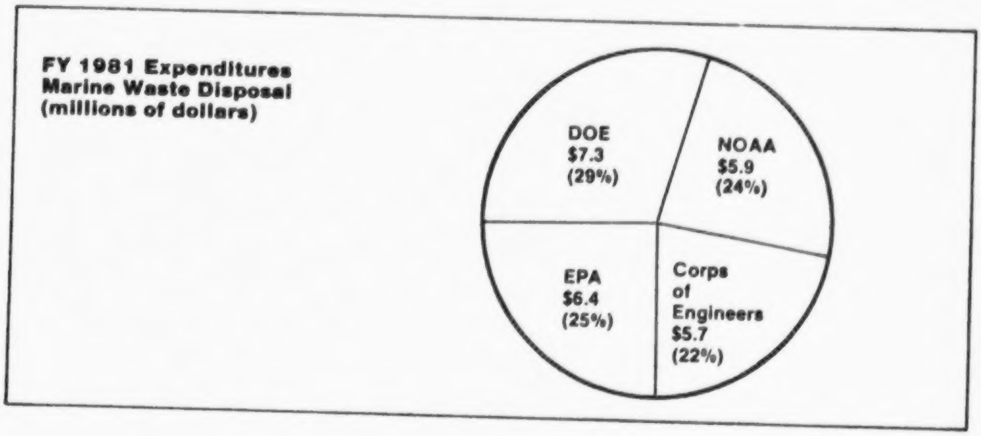
APPENDIX A

NATIONAL MARINE POLLUTION CONCERNS REGARDING MARINE WASTE DISPOSAL FROM THE NATIONAL MARINE POLLUTION PROGRAM PLAN: FEDERAL PLAN FOR OCEAN POLLUTION RESEARCH, DEVELOPMENT, AND MONITORING FISCAL YEARS 1981-1985 (INTERAGENCY COMMITTEE ON OCEAN POLLUTION RESEARCH, DEVELOPMENT, AND MONITORING, 1981)

MARINE WASTE DISPOSAL

Marine Waste Disposal is one of the most important areas in the Program. Some materials, such as dredged sediments, are now routinely disposed of in the oceans. Others, such as some types of radioactive wastes, may undergo ocean disposal in the near future. Pressure is growing to allow continued and increased disposal of sewage sludge in the oceans. For the purposes of this Plan, dredged material, industrial wastes, sewage wastes, radioactive materials, and brine discharges from Strategic Petroleum Reserve activities are considered under marine waste disposal. Each of these is discussed separately in the following sections. In addition, some general aspects of ocean waste disposal are presented in Chapter 5.

The principal agencies conducting research on marine waste disposal are the Department of Energy, the Environmental Protection Agency, the National Oceanic and Atmospheric Administration, and the Army Corps of Engineers (the Corps). The FY-1981 expenditures for waste disposal were spread relatively evenly among the four agencies as shown in the following figure.



Department of Energy funds were largely devoted to studies of brine disposal, and the Corps research focused on disposal of dredged material. Research funded by NOAA and EPA was more evenly distributed among the major types of wastes discussed in this section. The following table provides information on funds expended and percent of budgets allotted for research on each type of waste disposal in FY 1981.

MATERIAL DISPOSED	DOLLARS (Millions)	PERCENT OF WASTE DISPOSAL BUDGET	PERCENT OF TOTAL POLLUTION PROGRAM BUDGET
Dredged material	$9.3	36%	5.4%
Industrial waste	3.5	14	2.0
Sewage waste	4.4	17	2.6
Radioactive waste	2.2	9	1.3
Brine	6.0	24	3.5
TOTAL	$25.4	100%	14.8%

DREDGED MATERIAL DISPOSAL

Dredging of new channels and maintenance dredging of existing channels are required to provide safe and efficient navigation conditions for commercial and recreational marine transportation. Channel dredging generates significant amounts of dredged material consisting of the sediment and water mixture excavated from areas dredged. On the basis of volume, dredging is the largest single source of materials that are ocean dumped. During 1979, more than 72 million cubic yards of dredged material were deposited in the marine environment (COE, 1980). Of the 1979 total, 68% was disposed of in the Gulf of Mexico, 18% in the Atlantic Ocean, and 14% in the Pacific Ocean. The total 72 million cubic yards of dredged material constituted nearly eight times the combined tonnage of industrial wastes, sewage sludge, construction debris, and other waste materials disposed of in the marine environment during 1979 (CEQ, 1980). Compared with other materials that are disposed of in the ocean, most of the dredged material excavated in the United States is relatively innocuous, in many instances containing no harmful pollutants and, in most of the remaining cases, containing only trace levels of contaminants. In these cases, the primary concern associated with disposal of the relatively innocuous materials centers around the direct physical effects of disposal. These physical effects include burial of organisms, increased levels of suspended sediments, and accretion of disposed materials (COE, 1978). However, dredged material taken from highly polluted areas is usually contaminated with harmful chemical constituents such as heavy metals, synthetic organics, and oil and grease. Open-ocean disposal of these materials carries the threat of acute or chronic toxic effects on marine organisms, and potential contamination of human food resources. Much research has been conducted to describe the effects of dredged material disposal in the marine environment, and to evaluate disposal options that may be preferable to ocean dumping. A regulatory process based on the results of such scientific research has evolved to evaluate dredged material for ocean disposal, and to designate and monitor disposal sites.

Proposals for disposal of dredged material in freshwater and in coastal areas to the outer boundary of the territorial sea are regulated under the Federal Water Pollution Control Act (FWPCA). The jurisdiction of the Marine Protection, Research, and Sanctuaries Act (MPRSA) extends outward from the baseline from which the territorial sea is measured. Therefore, a zone of jurisdictional overlap exists between the baseline and the outer boundary of the territorial sea where, strictly speaking, the provisions of both the FWPCA and the MPRSA would apply. To eliminate this problem, EPA and the Corps have reached an agreement stipulating that only the MPRSA will be applied in the zone of overlap. Therefore, the vast majority of ocean-disposed dredged material must be evaluated under the MPRSA and pursuant regulations. The guidelines contained in the MPRSA take into account the international provisions of the London Ocean Dumping Convention. The intent of MPRSA, as it pertains to dredged material disposal, is to limit adverse ecological effects of ocean dumping. Title I of MPRSA stipulates that the Department of the Army reviews applications for permits and, when appropriate, issues permits for the transportation of dredged material to ocean disposal sites (Section 103). The Corps of Engineers has been designated by the Secretary of the Army to implement this authority. EPA designates ocean disposal sites and develops criteria for dredged material disposal. The Corps of Engineers reviews permit applications in accordance with the EPA criteria (Section 102). These review procedures are followed for applications to carry out private dredging operations. However, the bulk of dredged material is actually generated by Corps navigation projects, and MPRSA stipulates that the Corps must apply these same procedures and review criteria to disposal of materials generated by Corps projects. Title II of MPRSA requires that NOAA, in coordination with the EPA and the U.S. Coast Guard, conduct a continuing program of monitoring and research regarding the effects of ocean dumping of materials including dredged material.

In addition to the provisions of MPRSA and FWPCA, the Fish and Wildlife Coordination Act (FWCA) stipulates that Federal agencies involved in permitting or licensing dredging and dredged material disposal activities must consult with the F&WS and the National Marine Fisheries Service (NMFS) to ensure consideration of wildlife conservation.

The goal in regulating dredged material disposal is to allow dredging and disposal activities while avoiding unacceptable environmental risks. To ensure that the best practical procedures are employed in evaluating applications for ocean disposal of dredged material, the Corps of Engineers and EPA have worked jointly to develop a guidance manual for implementing Section 103 of MPRSA (EPA/COE, 1977). The manual summarizes and describes procedures for ecological evaluation of dredged material before ocean disposal. In overview, the permit evaluation and review process involves some or all of the following aspects: initial evaluation of sediment characteristics and potential for contamination; liquid, suspended-particulate, and solid-phase bioassay; evaluation of alternative disposal methods and justification for proposing ocean disposal; evaluation of bioaccumulation potential based on field or laboratory data; and general consideration of impacts on esthetics, recreation, economics, and other ocean uses. Recently a "matrix" approach has been applied on a trial basis in the New York District to assist in evaluating bioassay results. The purpose of the approach is to compare concentrations of specific pollutants (e.g., PCB, DDT, Hg) associated with dredged material with existing concentrations at the disposal site and at reference sites in the surrounding area. The Corps and EPA are jointly evaluating the "matrix" approach and other innovative guidance methodologies to improve the regulatory process.

Permits may be approved under MPRSA for disposal of dredged material at one of more than 100 EPA-designated interim sites. In 1979, 50 of the designated sites were actually used for dredged material disposal. The amount of dredged material that is disposed of in the ocean varies from year to year as a result of major new work or improvement navigation projects, weather patterns, and the fluctuating nature of maintenance dredging activities. Between 1973 and 1979, total volumes placed in ocean disposal sites ranged from 41 to 99 million cubic yards annually; no consistent time trends were evident during that period (CEO, 1980). In addition, more than 100 million cubic yards are dredged annually in coastal areas and disposed of by some means other than ocean dumping, e.g., upland disposal, beach nourishment (Samuels, 1981). Volumes of dredged material may increase because of future port improvements. On a regional level, disposal of dredged material from port improvements may have significant impacts. For example, it is anticipated that the improvements under consideration for Mobile and Norfolk would generate about 140 million and 35 million cubic yards, respectively.

Analysis and Conclusions

Dredged material disposal is recognized as a potential pollution threat in coastal areas. The severity of the threat can be controlled to a considerable degree by management of disposal activities. Research efforts conducted to date have concentrated on defining risks associated with dredged material disposal and evaluating the various disposal options, including ocean dumping. The principal Federal agencies involved in research on dredged material disposal are the Corps, EPA, and NOAA.

The Dredged Material Research Program (DMRP), conducted by the Corps of Engineers Waterways Experiment Station in Vicksburg, Mississippi, was completed in March 1978. The DMRP explored the physical and chemical impacts of dredged material disposal, and a variety of disposal alternatives, including beneficial uses of dredged material. The program was well planned, and DMRP results are thoroughly documented by technical reports on specific issues, synthesis reports, and summaries. At the planning and policy level, two major conclusions can be drawn from the results of DMRP (COE, 1978):

1) No single disposal alternative is preferable in all cases. Each dredging project is unique, and disposal options should be considered on a case-by-case basis.

2) An effective solution to the problem of dredged material disposal can be derived only from long-range regional planning and management of disposal activities.

The DMRP involved an expenditure of more than $32 million and covered a span of 5 years. The DMRP is the most comprehensive study performed on the disposal of dredged material and provides a large volume of background information along with the conclusions drawn from the engineering and scientific data accumulated during the study. Because of the limited time frame involved, 5 years, conclusions can be drawn only for the short term effects of the disposal of dredged material. To evaluate long-term effects, additional scientific and engineering data are needed. Therefore, the Corps of Engineers is continuing to collect data during FY 1982, although funds available to conduct the program are limited. Future Corps efforts will concentrate on those areas that relate to the long-term effects of dredging and disposal activities and will include the continuation of monitoring and evaluation of various parameters at selected DMRP field sites. Other ongoing or planned research activities of the Corps include monitoring and management studies conducted by the Corps district or division offices.

Past and ongoing EPA programs addressing dredged material disposal relate to the regulatory functions of the agency. Specifically, EPA, in conjunction with the Corps, is further developing bioassay techniques and innovative methodologies for evaluation of dredged material, researching the effects of PCB in dredged sediments, performing a field study at a dredged material disposal site, and conducting evaluations of potential dredged material disposal sites. Under the mandate of MPRSA, Section 201, the Department of Commerce, through NOAA, conducts a research and monitoring effort to improve our understanding of the effects of ocean dumped materials, including dredged material. Cooperative studies between NOAA and the Corps are being conducted at disposal sites in the New York Bight, near the mouth of Chesapeake Bay, and near the Mississippi River delta. These studies include baseline observations prior to dumping and long-term field and laboratory studies. Dredged material disposal is also being studied under the Long-Range Effects Research Program conducted under the mandate of MPRSA, Section 202.

Research or information needs related to the physical effects, chemical effects, and management of dredged material disposal have been identified. The needs and pertinent Federal activities are discussed below.

Physical effects on the ecosystem

This area is adequately addressed by ongoing Federal research programs. The disposal of dredged material may cause physical or chemical effects. Important examples of physical effects include burial of bottom-dwelling organisms, effects of increased turbidity and sediment type alteration, and the implications of changes in bottom topography. It is necessary to understand these effects to evaluate the implications of dredged material disposal.

This area is adequately addressed by ongoing Federal research programs. The DMRP has laid groundwork at the generic level for studying physical effects of dredged material disposal. Ongoing COE and NOAA programs are studying physical effects on a site-specific basis at several disposal sites. The generic aspects of physical effects have been adequately addressed. Site-specific studies performed on a limited basis continue to provide valuable information on long-term physical impacts.

Chemical effects on the ecosystem

The chemical effects of dredged material disposal become important when toxic or harmful constituents are present in dredged sediments. Research topics related to these contaminants include release and bioavailability, long-term fates, acute and chronic effects, and bioaccumulation in marine organisms including human food resources. In cases where dredged material is contaminated, information on the impacts of the contaminants is essential in evaluating disposal alternatives. Information on the effects of contaminants in dredged material is considered to be a most important need.

DMRP explored the generic aspects of dredged material contamination including contaminant release during and immediately after disposal, effects of contaminants, and bioaccumulation potential. Ongoing EPA, NOAA, and Corps of Engineers research and monitoring projects are also continuing to increase our understanding of chemical impacts.

This highly complex issue is not yet fully understood. Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future.' Studies of overall effects on biological communities and marine ecosystems should be emphasized to promote assessments of resource impacts. One way in which research on contaminated sediments might be advanced would be to carry out a synthesis and analysis of currently available information on the biological effects of contaminated sediment disposal. This would help describe the existing body of information, determine whether any conclusions can be drawn, and identify the most productive areas for future research. It is recommended that the value of such a study be determined by the Corps of Engineers, NOAA, EPA, and F&WS, and the study be conducted if appropriate. Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future.

Disposal management

Long-range management of dredged material disposal requires that basic information be available about potential disposal sites, estimated future volume and quality of dredged material, feasibility and cost of alternative disposal methods, prediction of impacts, and assessment of possible benefits of using innovative dredging technologies. In addition, disposal management requires development of a long-term strategy and provision of feasible disposal alternatives including ocean disposal. Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future.

Research, development, and monitoring related to this need fall largely within the purview of Corps of Engineers Headquarters, and coastal divisions and districts, and EPA and NOAA. In addition, the Corps and EPA are continuing to improve bioassay techniques for evaluation of dredged material prior to disposal.

There is a need for long-term regional planning for dredged material disposal. Information is required to develop regional management plans that would include projections of dredged material volumes and quality, rigorous analysis of the costs, risks, and benefits of the various disposal alternatives, development of several disposal options depending on the nature of dredged material involved, guidelines for selection of disposal options, and an assessment of the effects of the management program on ocean resources. More information is also needed to describe the cumulative effects of individual dredged material disposal actions on a regional and national basis. Where possible, evaluation of disposal impacts should be extended to the level of effects on fisheries stocks, recreational utility, and other ocean use impacts on a regional or national level.

INDUSTRIAL WASTE DISPOSAL

The manufacturing and processing techniques commonly used in our technological society convert raw materials into two types of outputs: products and wastes. Products are generally distributed and sold to consumers before final disposal; wastes may be processed and recycled, or, more commonly, disposed of in some other way. Industrial wastes may contain potentially harmful constituents including synthetic organics, heavy metals, and oil and grease. Commonly used disposal options for industrial wastes are secure landfills, ocean dumping, pipeline discharges to coastal and inland waters, and incineration. This discussion addresses the following industrial waste disposal options that involve the marine environment: ocean dumping, incineration at sea, and ocean outfalls.

Each of these activities differs from the others because of the type of discharge, the nature of the substances discharged, the level of treatment prior to discharge, or the regulatory procedures that apply. Therefore, background information on each of the three activities is presented separately below. However, research conducted on marine waste disposal is often pertinent to several disposal options. For example, basic fates and effects research conducted on a specific chemical constituent would provide useful information whenever that constituent enters the marine environment, regardless of the disposal technique used. To promote the application of general research to all three industrial waste disposal alternatives, a single group of research and information needs is presented in "Analysis and Conclusions," which follows the discussions of the options themselves.

Ocean dumping

When industrial wastes are ocean dumped, the waste materials are barged to a designated disposal site and discharged. The table below provides a summary of the volumes of industrial wastes that were ocean dumped from 1973 to 1979 (EPA, 1980b).

	INDUSTRIAL WASTE OCEAN DUMPED (Thousands of tons)						
Geographic Locations	1973	1974	1975	1976	1977	1978	1979
Atlantic	3,643	3,642	3,322	2,633	1,784	2,584	2,577
Gulf of Mexico	1,408	938	120	100	60	0.17	0
Pacific	0	0	0	0	0	0	0
TOTAL U.S.	5,051	4,580	3,442	2,733	1,844	2,548	2,577

In 1979, about 2,577,000 tons of industrial wastes were dumped in the ocean. There has been a trend toward reduction of total ocean dumping of industrial wastes since 1973 when ocean dumping became regulated by the Federal Government. Industrial waste dumping has been reduced by about 30% in the Atlantic, totally eliminated in the Gulf of Mexico, and was not practiced in the Pacific between 1973 and 1979. Three industrial-waste dumpsites were used in the Atlantic during 1979. These are the acid waste site in the New York Bight Apex, the Deepwater 106 Dumpsite located 106 nautical miles southeast of New York Harbor, and the Puerto Rico site located 40 nautical miles north of Puerto Rico. About 75% (by wet tons) of the industrial wastes ocean dumped in 1979 were acid byproducts of the titanium dioxide manufacturing process; the remaining 25% of the materials consisted of wastes from the manufacture of various chemicals including insecticides and pharmaceuticals.

Ocean dumping of industrial wastes is regulated under authorities assigned in the Marine Protection, Research, and Sanctuaries Act of 1972 (MPRSA), as amended. Under the legislation, transportation of industrial wastes for the purpose of dumping in ocean waters, the territorial seas, or the contiguous zone is prohibited except when authorized by a permit issued by the Administrator of EPA. Title I of MPRSA establishes a permit system and assigns to EPA the responsibility for review of permit applications and granting of permits, designation of dumpsites, and establishment of criteria to be used in reviewing permit applications. Title II of MPRSA assigns responsibility to the Department of Commerce (implemented through NOAA) to conduct, in coordination with EPA and the U.S. Coast Guard (USCG), a comprehensive and continuing program of monitoring and research regarding the effects of dumping materials, including industrial wastes, into ocean waters, coastal waters, and the Great Lakes.

Incineration at sea

The first incineration at sea of chemical waste officially sanctioned in the U.S. occurred in the Gulf of Mexico between October 1974 and January 1975 when M/T Vulcanus incinerated 16,000 metric tons of organo-chlorine wastes at a designated site about 140 nautical miles southeast of Galveston, Texas (EPA et al., 1980). In 1977, 17,600 tons of chemical waste were incinerated in the Gulf of Mexico, and 12,100 tons were incinerated in the Pacific Ocean (EPA, 1980b). Nearly all incineration at sea has been conducted under research permits granted by EPA under authority provided by MPRSA. From studies of these early burns, it has been concluded that incineration at sea for organic chemical wastes does not cause unacceptable environmental consequences, at least on a limited basis for some chemicals and at these specific sites (EPA et al., 1980). Candidate wastes for incineration at sea are generated primarily by the following industries: petroleum refining, organic chemical production, synthetic fibers and resins manufacture, and pesticides production.

Although incineration at sea has been conducted on a limited basis, it is likely to become more common in the future. Increasing amounts of industrial waste are produced each year, and increased regulatory pressure through implementation of the Resource Conservation and Recovery Act (RCRA) is eliminating some of the traditionally used land-based disposal options because of potential environmental hazards. The effects of RCRA implementation will be to prohibit inexpensive but potentially hazardous land disposal techniques, and to require costly site preparation, monitoring, and closure to ensure that industrial wastes are effectively isolated from the environment. As a result of these factors, RCRA implementation may indirectly promote incineration at sea, which will become more economically attractive as land disposal becomes more costly and highly regulated. Indicators of increasing pressure to promote incineration at sea are already evident. EPA (1980c) is considering designation of an incineration site in the North Atlantic. An *ad hoc* interagency work group has recently concluded that incineration at sea constitutes an environmentally acceptable and efficient means for destroying liquid, hazardous, organic chemical wastes (EPA et al., 1980).

Like ocean dumping, incineration at sea is regulated primarily under the authority of the Marine Protection, Research, and Sanctuaries Act and the London Ocean Dumping Convention. The responsibilities of Federal agencies are essentially the same as for ocean dumping.

Ocean outfalls

Industrial ocean outfalls are pipeline discharges of industrial wastes that directly enter estuaries, coastal waters, or oceans. Industrial wastes mixed with domestic and other municipal wastes are not specifically addressed in this section.

Ocean outfalls of industrial wastes are regulated by the EPA through the National Pollutant Discharge Elimination System (NPDES). In 1979, more than 5,000 NPDES permits were held for pipeline discharges by industries in coastal counties. In addition, about 7,500 operational discharges were associated with offshore oil and gas facilities. Pollutants that may be associated with various industrial effluents include synthetic organics, heavy metals, oxygen-consuming materials, suspended solids, and nutrients.

Authority for administration of the NPDES permitting procedure is assigned to EPA under Section 402 of the Clean Water Act. The Act requires that EPA establish limitations to effluent quality on an industry-by-industry basis. Specifically, for all point sources other than publicly owned treatment works, it is stipulated by the Clean Water Act that best practicable control technology currently available be attained by July 1, 1977, and best available technology economically achievable be attained by July 1, 1983.

In addition to the NPDES permit procedure described in Section 402 of the Clean Water Act, Section 403(c) of the Act requires the Administrator of EPA to develop ocean discharge criteria to be used in evaluating the effects of a discharge into the territorial seas, the contiguous zone, and the oceans. The provisions of Section 403(c) apply only to coastal and ocean waters and demonstrate the special concern of the Congress for the protection of marine water quality and marine resources. Final regulations under Section 403(c) were published on October 3, 1980, in the Federal Register 45:65942-65954.

Analysis and Conclusions

Determining the implications of industrial waste disposal in the oceans is difficult for several reasons. Many potential toxicants are present in these materials. In addition, composition of waste varies depending on the industry and the stage at which wastes are generated in the overall manufacturing process. To provide an extra measure of safety, wastes to be ocean dumped or incinerated are taken offshore, where dispersion and dilution can better mitigate adverse effects. However, dilution further complicates evaluation of pollution impacts because the focus shifts from acute impacts to sublethal, but possibly significant, effects that may occur in marine organisms, even at low concentrations of the discharged materials. Both acute and chronic effects on marine organisms must be understood, and risks to human health evaluated. Industrial outfalls are quite numerous in some areas and may adversely affect coastal ecosystems known to be highly sensitive to chemical and physical disruption, and essential to the maintenance of commercial marine fisheries, and recreational and aesthetic resources.

EPA and NOAA are the principal Federal agencies that conduct research and monitoring on ocean disposal of industrial wastes. EPA research is largely related to the regulatory functions of the agency. The Marine Disposal Research Program conducted by EPA includes an emphasis on industrial wastes. Research conducted involves a general assessment of human health risk associated with industrial waste disposal, development of the benthic microcosm as an analytical tool for predicting contaminant fates and assessing effects, and development of bioassay techniques for evaluating potential disposal sites. Information generated by EPA's Water Quality Research Program may also improve understanding of the effects of industrial waste disposal. These studies include investigations on the transport, transformation, and fates of toxic materials in marine ecosystems, studies on the effects of toxic materials on marine organisms, and evaluation of water quality criteria.

Within NOAA, the Ocean Dumping Program conducts research and monitoring on the effects of dumping waste materials in the ocean. Field research and monitoring are being conducted at the Deepwater 106 and Puerto Rico dumpsites to study the fate and effects of disposed industrial wastes. Complementary laboratory studies are also performed. In addition, the objectives of the Long-Range Effects Research Program are to determine potential long-range effects of ocean dumping on human health and ocean ecosystems, develop early warning systems to detect adverse changes in ecosystems, and improve understanding of basic ecosystem attributes and functioning. NOAA's Ocean Resources Coordination and Assessment Program is collecting inventory information on all pollution discharges, including industrial effluents, on a coast-by-coast basis (excluding the Great Lakes) with the following schedule for completion:

Gulf of Mexico - FY 1981
Alaska and West Coast - FY 1982
East Coast - FY 1983

In addition, the Microconstituents Program collects information on levels of contaminants in marine organisms.

The Food and Drug Administration, Department of Health and Human Services, also conducts a program relating to industrial waste. The Pesticides and Metals in Fish Program monitors and assesses contaminant levels in seafood.

Research or information needs related to the management of industrial wastes and the effects and risks associated with industrial waste disposal have been identified. The needs and pertinent Federal activities are discussed below.

Management of industrial wastes

Long-range planning and management of industrial waste disposal is essential to anticipate and mitigate pollution problems. Such planning requires that basic scientific and engineering information be available on future disposal requirements, options, risks, and costs. For example, the following tasks should be addressed:

- Identify wastes that will be generated on a national and regional level in the next 10 to 20 years.
- Characterize constituents of wastes and assess the potential for adverse environmental effects under various disposal options.
- Determine feasibility, environmental risk, and economic cost of the various disposal options, including land-based options and recycling.
- Develop long-range management strategies that would indicate preferred options for various wastes, potential sites for disposal, and any regulatory changes that would be appropriate.

Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future.

EPA regulatory programs conducted under the mandates of the Marine Protection, Research, and Sanctuaries Act, and the Clean Water Act, and coordinated activities in EPA's Office of Research and Development are now addressing this need. In addition, NOAA research programs provide basic information that can be used in predicting the fates of disposed materials and their effects in marine ecosystems. However, these issues are complex, and in many cases detailed site-specific analyses are required. Support for research and development that will provide information needed for management of industrial wastes should be continued. In addition, individual industries in the private sector should be encouraged to characterize and evaluate their respective waste products. States and local governments should work together to implement management strategies.

Ecosystem effects & human health risks

Waste disposal management decisions must be based on an understanding of the fates of contaminants in marine ecosystems, their effects on organisms, and the pathways by which humans may be exposed to contaminants. Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future. Future emphasis is required in the following areas:

- Fates and transformations of disposed materials.
- Bioaccumulation of contaminants in commercial and recreational marine species.
- Human risk assessment models including health risks associated with various waste constituents, containment levels in human food resources, and consumption patterns in various human populations.
- Effects on marine ecosystems, especially in sensitive coastal areas.
- Significance of chronic sublethal effects in marine organisms (e.g., mutagenicity, community alteration, reduced fecundity).
- Effects on fisheries yields.
- Field verification of laboratory results.

Most of these issues are under study by EPA or NOAA. However, the current level of effort in EPA is now limited by availability of funding; it is estimated that in FY 1981 less than $600,000 was expended on industrial waste disposal research under EPA's Marine Disposal Research Program. The acquisition of information on the fates and effects of ocean disposal industrial waste could be accelerated by (1) improved coordination and communication between managers of NOAA and EPA research programs, and (2) increased funding to EPA's Marine Waste Disposal Program.

SEWAGE DISPOSAL

Disposal of the various types of wastes generated by the densely populated coastal areas has created economic, environmental, and political conflicts. Historically, the nation's rivers, estuaries, and coastal waters have received municipal waste discharges since collection and treatment of domestic wastes was initiated. Prior to the 1970's, ocean disposal was largely unregulated, and adverse impacts on human health and the environment were observed. The principal hazards to human health from sewage waste disposal are associated with the transmission of human pathogens and the ingestion of seafoods contaminated with toxic metals and synthetic organic compounds. In the Great Lakes there are additional risks from pathogens and toxics because these waters are used as a source of drinking water. Food poisoning, dysentery, and transmission of a variety of human and animal parasites are commonly associated with the discharge of untreated sewage waste. Other adverse impacts are the loss of recreational and commercial resources where beaches or wetlands are fouled with floating waste materials or are closed to fishing and shellfishing because of sewage contamination.

Public concern over the pollution of coastal waters during the 1970's precipitated the enactment of two major legislative measures aimed at improving the quality of the marine environment. The Marine Protection, Research, and Sanctuaries Act of 1972 (P.L. 92-532), also known as the Ocean Dumping Act, was enacted to regulate ocean dumping of all materials and prevent or strictly limit the dumping of any material that would adversely affect human health, welfare, amenities, the marine environment, ecological systems, or economic potentialities. The Act was further amended in 1977 mandating that the ocean dumping of harmful sewage sludge cease by December 31, 1981 (NACOA, 1981).

The Federal Water Pollution Control Act, also known as the Clean Water Act, regulates, among other things, municipal waste disposal by pipeline discharge into the oceans. The Act was substantially amended in 1972 (P.L. 92-500) to establish a single standard for the nation as a whole by requiring all publicly owned treatment works (POTW) to achieve by 1983 an effluent water quality based on secondary treatment. Exemption from the secondary treatment mandate for coastal POTW dischargers on a case-by-case basis was later provided under Section 301(h) of P.L. 95-217 in the 1977 amendments. The EPA regulations developed under the Clean Water Act also require compliance with State water quality standards.

EPA authority under the Ocean Dumping Act includes review, award, and enforcement of ocean dumping permits, designating and managing disposal sites, and developing criteria to evaluate ocean dumping permit applications. EPA's posture has been, until recently, that any practical (i.e., feasible and available) disposal alternative is preferable to ocean dumping. However, on the basis of new information about environmental effects and economic costs of ocean dumping and its alternatives, and the increasing rate at which sewage wastes are generated at the national level, the EPA is reevaluating its restrictive position (EPA, 1981).

During the 1970's EPA, under its ocean dumping permit program, developed stringent tests and criteria to restrict ocean dumping. The environmental impact criteria require that the candidate material pass two bioassay tests: a test for toxicity of the liquid, suspended particulate, and solid phases; and a bioaccumulation test. Failure of either test results in permit denial, except for emergency or research permits. However, owing to problems in the testing procedure, the results of these tests have been ambiguous.

EPA's permit program authorized by Section 402 of the Clean Water Act establishes limits on pollutants that can be discharged from municipal point sources through outfalls, including sludges and wastewater effluents, into marine waters. The factors used to evaluate whether a discharge will cause "unreasonable degradation" of the marine environment include the chemical constituents of the discharge, their potential for causing adverse impacts on the environment and to human health, the sensitivity and importance of the biological community, and the impact on special aquatic environments and fisheries. Upgrading of POTWs to secondary treatment as required by this Act will significantly improve the quality of wastewater effluents but will result in the generation of greater volumes of sewage sludge.

The Ocean Dumping Act prohibits ocean dumping of harmful sludge after December 31, 1981, and EPA regulations under the Clean Water Act will prohibit the disposal by outfall pipes after July 1985. However, there is mounting evidence that the environmental effects and costs of other methods of waste disposal would exceed those of disposal in the oceans, and the scientific information available to date does not justify the total ban on ocean disposal of harmful sewage sludge (NOAA, 1979a). The feasibility of land disposal is limited by shortage of available land near urban centers and the threat of pollution to groundwater resources. Recycling of sludge is possible, but expensive. Tertiary treatment would cost substantially more than the currently used treatment levels. For all of these alternatives, sludge solids remain and would have to be disposed of on land unless they were incinerated. Incineration could contribute significantly to the air pollution problems already existing in many heavily populated areas.

Sewage effluents that have been treated still contain substantial quantities of suspended solids, toxic metals, synthetic organic compounds, fecal coliforms, and other potentially pathogenic microorganisms. Effluents also contain oxygen-demanding organic substances, and various forms of the nutrients nitrogen and phosphorus (NACOA, 1981). Secondary sewage treatment is moderately effective in reducing the number of pathogenic microorganisms. Combined primary and secondary treatment reduces the fecal coliform level in the effluent. Discharge of the effluents through outfall pipes is regulated by the EPA under Sections 402 and 403 of the Clean Water Act (as amended). Sewage effluents are discharged directly into the oceans through outfall pipes, primarily by Los Angeles, New York, Boston, and Miami.

Sludge is a byproduct of sewage treatment. In primary treatment plants, floatable materials are removed and solids are settled to produce the sludge. In a secondary treatment plant, bacteria are employed to decompose dissolved and colloidal organic matter from the wastewater stream, and the bacterial debris becomes part of the sludge. Generally, a portion of secondary sludge is anaerobically digested, a process that decomposes much of the solid organic matter to gaseous byproducts such as methane. This process reduces the coliform and viral content of the sludge to very low levels. The chemical composition of sewage sludge is highly variable and depends on the content of receiving wastes, the level of treatment, and operating efficiency of plants. POTWs receive wastewater streams from a variety of sources, including residences, businesses, small industries, and even major industries in some instances. Many cities have combined sewer systems, with rainwater and street runoff entering the POTWs along with sanitary wastes. During the treatment process, the fates of particular constituents, organic and inorganic, are dependent upon the specific physical, chemical, and biological conditions of plant operations. Generally, however, the sludge concentrates toxic metals (cadmium, copper, lead, mercury, and zinc), pathogens, organic matter, petroleum hydrocarbons, and synthetic organic compounds such as polychlorinated biphenyls (PCBs).

About 6 million dry metric tons of sewage sludge are produced annually by the 14,500 POTWs in the United States. Sewage sludge generation is projected to increase to about 10 million dry metric tons per year by 1990, after all POTWs have converted to secondary treatment. In the United States sewage sludge is disposed of by land application (20%); landfill (40%); incineration (residual ash landfilled) (25%); and ocean disposal (15%) (Vaccaro et al., 1981). In 1979, 5.9 million wet metric tons of sewage sludge were ocean dumped, and 2.8 million wet metric tons were discharged through the

Los Angeles and Boston outfalls. (As a rule of thumb, wet sludge may be assumed to consist of 90% water and 10% solids.) Discharges through outfalls are regulated by EPA under Sections 403 and 405 of the Clean Water Act and those by ocean going vessels under Section 102 of the Ocean Dumping Act.

Under Title II of the Ocean Dumping Act, NOAA has the lead responsibility for conducting comprehensive research and monitoring on the fates and effects of ocean dumping. NOAA has concentrated its research and monitoring efforts at the 12-mile dumpsite in the New York Bight, where much of the dumping occurs, at the 106-mile dumpsite, which is primarily used for industrial wastes, and at the the recently deactivated Philadelphia sewage sludge dumpsite. The efforts include baseline observations prior to dumping, experiments during dumping, and long-term field and laboratory observations. Studies at the Philadelphia dumpsite continue to describe any changes that occur following the cessation of dumping. NOAA's region-specific research programs to determine the fate and effects of pollutants such as sewage include the Hudson-Raritan Estuary Project, the New York Bight Project, Puget Sound Project, the Great Lakes Pollution Studies, and the Habitat Investigations Program.

Analysis and Conclusions

Sewage effluents and sewage sludge are recognized as pollutants and pose a potential threat to the marine environment. However, other sources of contaminants also contribute to the pollution of the coastal waters. The GAO estimated that less than one-half of the pollutants entering the waters is from municipal treatment plants and other regulated point sources (CEO, 1980). The remaining amount is contributed by the combined discharges of estuaries and rivers, direct runoff from nonpoint sources, atmospheric fallout, and ocean dumping (NOAA, 1979a).

EPA's effort related to the ocean disposal of sewage is focused on evaluation of ocean dumpsites and studying the impacts of discharges from ocean outfalls. Under EPA's Dumpsite Evaluation Program, conducted by the Office of Water and Waste Management, ocean dumping sites are being evaluated. Evaluation consists of biological, water, and sediment surveys followed by assessment of the environmental effects that would result from use of the dumpsite.

The Ocean Outfall Research Program (Municipal Waste Disposal) deals with the fate and effects of the discharge of wastewater effluents. The research involves assessment of the impacts associated with municipal wastewater discharges into estuarine and marine environments. Emphasis is placed on the following areas: marine food chain contamination by toxic materials; transport and transformation of pollutants in the marine environment; assessment of human risk associated with the consumption of contaminated seafood; and surveys of sediment quality and benthos contamination near municipal outfalls. In addition, studies on ecosystem level effects of wastewater pollution (and pollution abatement), monitoring techniques (including bioassays), and site-specific outfall studies (including the use of remote survey techniques) are being conducted.

EPA is studying also sewage treatment and sludge disposal options not related to ocean disposal, such as landfill applications, incineration, and composting. These may be considered alternatives to ocean disposal of sewage wastes.

Research and information needs related to the effects of ocean outfalls, the effects of ocean dumping, and management of sewage wastes have been identified. The needs and pertinent Federal activities are discussed below.

Effects of sewage outfalls

More information is needed on the fates and effects of materials discharged through outfalls to describe the implications to marine populations including long-term sublethal effects on aquatic species, and human health risks. Full understanding is needed of marine food webs and the movements of various waste constituents through marine ecosystems. Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future.

EPA's Municipal Waste Program provides basic information needed to implement Section 301(h) of the Clean Water Act, which allows relaxation, under certain conditions, of the requirement for secondary sewage treatment. Research is conducted on marine food chain contamination by toxic materials, transport and transformation of pollutants, human health risks associated with seafood consumption, and sediment quality and contamination of benthic communities near outfalls.

NOAA conducts research to determine the fates and effects of human-induced and natural changes on the abundance, distribution, and functioning of living marine resources, assess the health of fishery stocks, and assess risks to human health associated with ingestion of contaminated seafood.

Present research programs conducted by EPA and NOAA should continue. In addition, EPA should perform a review and synthesis of the statistical data resulting from site-specific studies conducted by states in conjunction with Section 301(h) waivers. EPA should use the review to determine the overall value and effectiveness of secondary municipal waste treatment, and of the 301(h) exemption program. The review should be completed by the end of 1984. To accomplish this, planning for the review should be initiated in FY 1982.

Effects of sludge dumping

Sewage sludge contains substances that are potentially harmful to living marine resources, ecosystems, and human health. More information is needed to describe changes that occur in an ecosystem after dumping is stopped and to determine the impacts on the immediate dumping area and how far from the dumping site impacts occur. In general, this area is adequately addressed by ongoing Federal research programs.

NOAA's Ocean Dumping Program includes baseline observations prior to dumping, experiments during dumping, and long-term field and laboratory observations. This program is also laying the groundwork for determining the changes in the marine ecosystem following cessation of sludge dumping at the recently deactivated Philadelphia dumpsite.

These are complex issues that have not been fully addressed. NOAA and EPA should continue research to determine the effects of dumping. Continuing and improved coordination between EPA and NOAA is required. EPA should focus on case-specific studies directly related to the regulatory process, and NOAA should conduct research to obtain a more general understanding of natural and altered marine ecosystems, with the expectation that dumping will continue or resume at certain sites. Studies should focus on ecological effects beyond the designated dumpsites. In particular, research should be conducted on the pathways and fates of key pollutants such as PCBs and heavy metals.

As dumping of sludge is discontinued at a specific site, NOAA should conduct monitoring; research activities to obtain information on the changes in the marine ecosystems should continue. NOAA's study of the abandoned Philadelphia dumpsite is under way.

Management of sewage wastes

Long-range management of sewage disposal requires multimedium assessments including the environmental impacts, economic costs, and energy consumption of disposal methods including land application, landfill, land and ocean incineration, and ocean disposal using barges or outfalls. Specifically, the following actions are needed:

- Determine the volumes of sewage to be generated in coastal areas during the next 20 years.
- Compare the disposal options, such as ocean dumping or outfall disposal of sludges.
- Quantify cost and describe effectiveness and environmental gain of the different levels of treatment technologies.

- Conduct special environmental and economic studies related to Section 301(h) exemptions.
- Select the most appropriate ocean disposal sites.

Although this area is now addressed by ongoing programs, it is of continuing importance and should receive emphasis in the future. The EPA has only recently begun a program to compare directly the environmental risks and costs of various methods of land disposal, incineration, and ocean disposal.

Higher levels of treatment result in cleaner effluent discharges while producing larger quantities of sludge containing concentrations of heavy metals. Decisions have to be made to determine the most appropriate level of treatment, the preferred method of waste disposal, and the best site for disposal.

Although planning and implementation of sewage disposal programs are the responsibility of State and local governments, EPA and NOAA should assist in determining the effects of various ocean disposal options. The following require special attention: (1) determining the chemical composition of wastes entering POTWs in order to remove contaminants through treatment; (2) determining the relationships among treatment type, effluent quality, environmental effects, and human health effects; and (3) selecting the ocean disposal sites that are the least disruptive to the marine environment.

RADIOACTIVE WASTE DISPOSAL

For the past three decades, radioactive wastes have been produced by a variety of activities in the United States. Low-level wastes generated by medical, industrial, and research activities contain low-level but potentially harmful quantities of radionuclides. High-level waste is produced by our national defense programs and generally requires shielding and long-term isolation from the environment. Substantial quantities of both high- and low-level wastes are generated by the nuclear power industry. There is no method of reducing the time that a particular substance remains radioactive; therefore, the wastes must be effectively isolated from the environment until they become harmless. While the medical use of radioactivity can be beneficial to man, uncontrolled exposure may be harmful. Because of the increasing volumes of wastes generated by the various activities, waste disposal management has become an urgent national concern. Safe disposal of the waste produced by the nuclear power industry is one of the factors affecting the industry's growth (NOAA, 1980b). A thorough understanding of the fate and effects of these materials in the marine environment is required to evaluate radioactive waste disposal options.

On February 12, 1980, culminating nearly 2 years of technical and policy review, the nation's first comprehensive program for managing radioactive waste was announced. Under this program, priority efforts for the disposal of high-level radioactive waste will be focused on land-based mined repositories with the use of emplacement in stable ocean sediments as a longer range alternative. The Department of Energy (DOE) will have lead responsibility for planning the non-regulatory waste management program and interfacing with the regulatory agencies, i.e., the Environmental Protection Agency (EPA) and the Nuclear Regulatory Commission (NRC) (Congressional Record, 1980).

Because low-level and high-level radioactive wastes are fundamentally different in character and in disposal requirements they are discussed separately. Three study needs of major importance are then identified as subjects for Analysis and Conclusions.

Low-Level Waste Disposal

Between 1946 and 1970 the Atomic Energy Commission (AEC) licensed the dumping of more than 86,000 containers of low-level radioactive wastes at 28 recorded dumpsites in the Atlantic and Pacific Oceans and the Gulf of Mexico. The Farallon Islands sites received approximately 97% of the radioactive materials dumped in the Pacific Ocean. Approximately 96% of the recorded radioactive metals dumped in the Atlantic were received in two sites more than 100 miles off Sandy Hook and approximately 150 miles apart. Only two dumps were made in the Gulf of Mexico (EPA, 1980e).

In 1960, the AEC placed a moratorium on the issuance of new licenses for at-sea disposal of nuclear waste and designated two land locations at Idaho Falls, Idaho, and Oak Ridge, Tenn., as interim low-level waste burial sites for AEC licensees. In 1962, the first permanent commercial disposal site on land for low-level radioactive waste (in Nevada) was licensed and available for use by private organizations. Shortly thereafter, licensed facilities for commercial land burial were established in Illinois, Kentucky, New York, South Carolina, and Washington. As land burial facilities became available, AEC stopped issuing new licenses to commercial firms to collect radioactive waste to be dumped. Between 1965 and 1970 only a small amount of low-level radioactive waste was dumped, largely because land disposal is cheaper. Capacities have been reached at three of the six commercial sites, and only three sites remain open. In addition, the DOE has 14 active and 2 closed land burial sites (NSF, 1971).

Ocean dumping was discontinued in June 1970 following a policy recommendation by the President's Council on Environmental Quality (CEQ) in its 1970 report to the President. This report recommended that low-level wastes be dumped only when "no practical alternative offers less risk to man and his environment" (CEQ, 1970). This policy was incorporated into AEC regulations and, in 1972, was included in the Ocean Dumping Act. The Ocean Dumping Act, among other things, designates the EPA as the agency responsible for issuing ocean disposal permits and for environmental guidelines and standards applicable to ocean disposal of radioactive waste.

High-Level Waste Disposal

High-level radioactive wastes are generated by facilities that reprocess irradiated nuclear reactor fuel. Spent nuclear reactor fuel assemblies, if discarded, are also high-level wastes. Materials contaminated with transuranic elements from the reprocessing of reactor fuels and the fabrication of plutonium to produce nuclear weapons also require long-term isolation methods. High-level wastes are temporarily stored at reactor sites and at Federally managed sites in Washington, South Carolina, and Idaho. The United States has never disposed of its high-level wastes in the oceans, and in 1972 two major legislative initiatives were enacted prohibiting future disposal of high-level wastes into coastal waters and rivers. The Ocean Dumping Act, in addition to regulating ocean disposal of low-level waste, prohibits the dumping of high-level waste and radiological warfare agents in ocean waters. Soon after the Ocean Dumping Act was enacted, the Clean Water Act was amended to extend the prohibition to all navigable waters. Although not immediately contemplated, subseabed emplacement of high-level radioactive wastes is a future option. Present international conventions preclude the dumping of high-level wastes in the oceans; it is uncertain at this time, however, if these conventions extend to the emplacement of high-level wastes in the geologic formations of the subseabed.

Analysis and Conclusions

In 1971, the National Academy of Sciences (NAS) concluded that, in terms of ecological effects, the consensus of the scientific literature was that radionuclides are not likely to be significantly deleterious to populations of marine organisms at the dose rates estimated for the most contaminated environments. Although the NAS predictions are subject to revision in the light of new knowledge, there has been no evidence to date that past practices for radioactive waste disposal in the oceans have jeopardized the health of humans or any marine species (Dyer, 1976; Hawkins, 1980).

In 1978, scientific experts reviewed the status of our scientific knowledge on all ocean pollutants and recommended areas where further study was necessary. These experts concluded that to date, no impacts on human health have been documented from the ocean disposal of radionuclides and no effects harmful to marine organisms are known, even at the sites of large discharges. These experts did recommend, however, that existing dumpsites be monitored for leakage of radionuclides to test the validity of present assumptions about the retention of disposed materials in the sediments, and to provide a basis for the selection of potential future disposal areas for low-level wastes (NOAA, 1979f).

Research and information needs related to existing low-level disposal sites, management of low-level wastes, and the possible subseabed disposal of high-level wastes have been identified. The needs and pertinent Federal activities are discussed below.

Effects at disposal sites for low-level radioactive wastes

Existing disposal sites provide an excellent experimental situation to study the physical, chemical, and biological processes that incorporate, transform, and accumulate radioactive elements and cause these toxic substances to migrate from the disposal canister to biological receptors (including humans). In general, this area is adequately addressed by ongoing Federal research programs.

The EPA supports research to evaluate problems and limitations associated with ocean disposal as one alternative in a low-level radioactive waste management program. The objectives of the program are to determine the fate and behavior of the radioactive waste packages that were dumped in the Pacific and Atlantic Oceans between 1946 and 1970 so that predictions of future environmental impact can be made if use of the ocean disposal alternative is again contemplated.

Since September 1980 EPA and NOAA have been discussing the areas where NOAA could assist EPA in its study of the long-term impacts of low-level ocean disposal and have jointly developed a monitoring plan. EPA and NOAA plan to enter into a memorandum of understanding by September 1981 to implement the monitoring plan.

Studies undertaken by EPA and NOAA should employ existing disposal sites to determine release rates of radioactive materials to sediments and water, to detect uptake by organisms, particularly sedentary species, and to identify bioaccumulation processes. Monitoring programs should be designed to detect any physiological or morphological abnormalities in resident biota and to identify in situ conditions where more subtle physiological processes involving radionuclides might be studied.

Management of low-level radioactive wastes

A disposal management strategy based on social, environmental, and economic factors is needed to determine the appropriate technique for low-level waste disposal on land and in the oceans. In general, this area is adequately addressed by ongoing Federal research programs.

The EPA's Dumpsite Evaluation Program is considering sites for potential marine disposal of low-level wastes. Previously used radioactive dumpsites have been surveyed to assist in developing criteria for selection of potential disposal sites and for the development of practices for monitoring future dumpsites.

Environmental risks of subseabed disposal of high-level radioactive wastes

Assuming the environmental and technical feasibiity, the technology for subseabed disposal will be another 10 years, at least, in developmental stages. Under present national and international laws, ocean dumping of high-level radioactive wastes is prohibited. However, on both the national and international levels, the subseabed disposal option for emplacement in submarine geologic formations is being considered as a potential technical alternative.

The Subseabed Disposal Program focuses on an investigation of the environmental and technical feasibility of subseabed disposal of high-level radioactive wastes. This program is considered a long-range concept for waste isolation and is still in the evaluation stage. The program's nearterm objective is to determine whether the deep-ocean sediments are effective barriers for confinement of wastes emplaced within geologically stable and biologically inactive regions of the deep-ocean floor.

Although the subseabed disposal of high-level radioactive waste would not occur, at the earliest, until after 1995, decisions to employ this disposal option will probably be made much earlier. Environmental risks must be assessed before decisions to dispose of high-level wastes in the subseabed can be made. Basic information is needed on the stability of deep-sea sediments when exposed to different temperature conditions that might exist in the vicinity of high-level waste disposal canisters. Information on sediment characteristics is also essential in order to evaluate different emplacement technologies. A better understanding of the advection and diffusion characteristics of deep ocean waters and the migratory patterns of deep-sea animals is important for identifying potential pathways of radioactive material back to man. This area is adequately addressed by ongoing Federal research programs.

DOE and NOAA have under consideration a memorandum of understanding for cooperation to investigate the feasibility of seabed emplacement. NOAA's involvement and assistance could be in activities such as site surveying and charting of the ocean floor (using the oceanographic fleet), assessing engineering emplacement methods, reviewing the characteristics of deep-water biological communities, reviewing data on ocean sediments as geological barriers, using the satellite system to transmit data, and assessing pollution on site.

- During the period of time covered by this Plan (through FY 1985), DOE and NOAA should undertake activities to designate environmentally acceptable and geologically stable disposal sites. These joint activities should include the charting of the ocean floors at and in the vicinity of potential disposal areas, reviewing data and analyses of sediment samples to assure that the sediments are effective barriers for confinement of the waste materials, and assessing the data on biological communities at the proposed disposal sites.

BRINE DISPOSAL

A concentrated brine solution needing disposal results from the development and operation of salt dome storage cavities associated with the Strategic Petroleum Reserve Program (SPRO). Mandated by the Energy Policy and Conservation Act of 1975 (P.L. 94-163), SPRO is intended to help protect the United States from severe disruptions in the world oil supply by stockpiling crude oil. To store crude oil, cavities are formed in salt dome structures by injecting freshwater or seawater, which forms a concentrated salt solution by dissolving a portion of the solid salt structure. After alternating injections of water and withdrawals of saturated brine, the cavity is completed in which crude oil may be stored. The storage area is equipped with separate pipes to the top and bottom of the cavity. Crude oil pumped in through the top pipe rides above the saturated brine, which is denser than the oil. The addition of oil to the top layer forces brine up and out through the pipe at the bottom of the cavity. Crude oil is extracted by pumping seawater to the bottom of the cavity, thus forcing oil out through the pipe at the top of the cavity.

All SPRO salt dome storage sites are located along the coast of the Gulf of Mexico from Texas to Louisiana. Sixteen dome cavities were available to SPRO because of previous salt dome storage by the chemical and petroleum industries. Some existing cavities will be used as is, some will be expanded, and a limited number of new cavities will be created to provide the total volume needed. Large volumes of brine at 280 parts per thousand, about eight times the concentration that occurs in full-strength seawater, are produced by the process of cavity formation. For example, to mine a new 100-million-barrel storage cavity, a volume of about 700 million barrels (111 billion liters) would be injected as seawater and extracted as brine. During the routine operation of filling the cavity with crude oil, a volume of saturated brine equal to the volume of emplaced oil would normally be discharged. Seawater injected to effect removal of crude oil dissolves salt from the cavity walls, causing storage area growth as a byproduct of crude-oil removal. This limits the number of fill cycles a storage cavity can undergo. SPRO engineers generally assume a cavity lifetime of five cycles over a period of at least 25 years (NOAA, 1980c). Currently, about 160 million barrels of crude oil are stored under SPRO (DOE, 1980). It is planned that by 1989, 750 million barrels of oil will be stored and maintained in Gulf Coast salt dome cavities.

Disposal of large volumes of nearly saturated brine solution is the primary marine environmental problem associated with SPRO. The brines are normally discharged into the Gulf of Mexico through diffusers to promote rapid dilution to background salt concentrations. However, the brines are denser than seawater and tend to flow along the sea floor, affecting bottom-dwelling organisms until sufficient dilution has been achieved. In addition, hydrocarbons and other harmful constituents may be dissolved in or entrained with the brine discharge. Brine discharges are regulated by the EPA under the National Pollution Discharge Elimination System (NPDES).

Analysis and Conclusions

To assess the environmental implications of brine discharges, the fate of brines should be predicted and monitored, and effects on marine ecosystems studied. In addition, any other potentially harmful chemical constituents found in brine require special studies.

A substantial research and monitoring program was initiated in 1977 to study the environmental implications of SPRO brine discharges. The program is funded by DOE and managed by the Environmental Data and Information Service (EDIS) in NOAA. Between 1977 and 1982, about $14.5 million was invested in modeling the fate of discharged brine and studying the effects of the discharges in the Gulf of Mexico. In general, the results of the studies indicate that the environmental effects of SPRO brine disposal are of relatively little consequence in most cases. In the future, environmental studies and monitoring are planned to continue, but possibly at a reduced level.

Information needs in this area are considered to be of lesser importance in the context of national marine pollution concerns. The interagency program funded by DOE adequately addresses the needs related to environmental implications of brine disposal. Results of earlier studies appear to justify a reduction in the intensity of these studies; information needs have been met.